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Ecosystem Services and Biodiversity

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Introduction

Ecosystem Services and Biodiversity

Ecosystems provide a multitude of benefits to humanity, from food, clean water and flood protection to cultural heritage and a sense of place, to name but a few. However, many of these benefits, known as 'ecosystem services', are under severe threat from man-made pressures. Decision makers need clear information on how biodiversity underpins these services, the demand for them, the capacity of ecosystems to provide them and the pressures impairing that capacity. In this report we explore four core facets of the ecosystem services concept: the links between biodiversity and ecosystem services; current techniques for mapping and assessing ecosystems and their services; valuation of ecosystem services and the importance of considering all ecosystem services and biodiversity as part of an interconnected system.

Relaxing on apple tree. ©iStock.com/Mac99

Habitat degradation, over-exploitation, invasive alien species, pollution and climate change are all affecting ecosystems across the globe (Pereira, Navarro & Martins, 2012; Barnosky *et al.*, 2011). It is estimated that 60% of the world's ecosystems are degraded or used unsustainably; 75% of fish stocks are over-exploited or significantly depleted and 13 million hectares of tropical forests are cleared each year (MA, 2005; UN FAO, 2011). Loss of biodiversity is proceeding at such a rate that we may face a mass extinction event if trends continue (Barnosky *et al.*, 2011).

Biodiversity decline represents not only an irreversible loss to the planet but also threatens humanity's life support system: the services that nature provides represent everything from the food we eat to the air we breathe (Díaz *et al.*, 2006; Cardinale *et al.*, 2012; Hooper *et al.*, 2012).

What are ecosystem services?

Ecosystem services are the many different benefits that ecosystems provide to people (MA, 2005). For example, a stand of trees can reduce air pollution, purify the water supply, reduce the likelihood of floods and help regulate the climate by capturing and storing carbon. It might also provide timber for buildings, a space for recreation and improve the aesthetic qualities of the landscape.

Despite the importance of these services to people, in the past many have been taken for granted, being viewed as free and infinite. However, it is now clear that the worldwide degradation of ecosystems is also reducing the services they can provide (MA, 2005). The ecosystem services concept provides a starting point towards defining, monitoring and valuing such services. Making the fundamental nature of these services explicit not only helps to raise awareness of the importance of protecting ecosystems, it can also provide decision makers with quantitative data, enabling them to consider all aspects of the socio-economic-ecological system in which we live (see Figure 1 on page 4). In this way we can work towards policies which protect biodiversity while optimising sustainable use of ecosystems, allowing both humanity and ecosystems to thrive.

> *'Despite the importance of [ecosystem] services to people, in the past many have been taken for granted, being viewed as free and infinite.'*

The rise of the ecosystem services concept

The concept of ecosystem services was brought into widespread use by the Millennium Ecosystem Assessment (MA), a global initiative set up in 1999 to assess how ecosystem change would affect human well-being (MA, 2005). The MA defines ecosystem services simply as:

"the benefits that people obtain from ecosystems". This encompasses both goods, such as timber, and services such as air purification. The MA divided these services into four categories:

i. *Supporting services.* These are services, such as nutrient cycling and soil formation, which are needed for the production of all other services.

ii. *Provisioning services.* Products obtained from ecosystems, such as food or timber.

iii. *Regulating services.* The benefits obtained from the regulation of ecosystems, including services such as purification of water, flood control, or regulation of the climate via carbon sequestration.

iv. *Cultural services.* The benefits people obtain from ecosystems through spiritual enrichment, cognitive development, reflection, recreation, and aesthetic experiences.

Following the MA, The Economics of Ecosystems and Biodiversity (TEEB) initiative was launched in 2007. Centred on economic valuation, TEEB aims to help decision makers recognise the economic benefits of biodiversity and the growing cost of ecosystem degradation (TEEB, 2010).

In Europe, in 2011, the European Commission adopted the Biodiversity strategy to 2020. Target 2 of the strategy aims that *"by 2020, ecosystems and their services [will be] maintained and enhanced"* and to achieve this, Action 5 of this target foresees that Member States will *"map and*
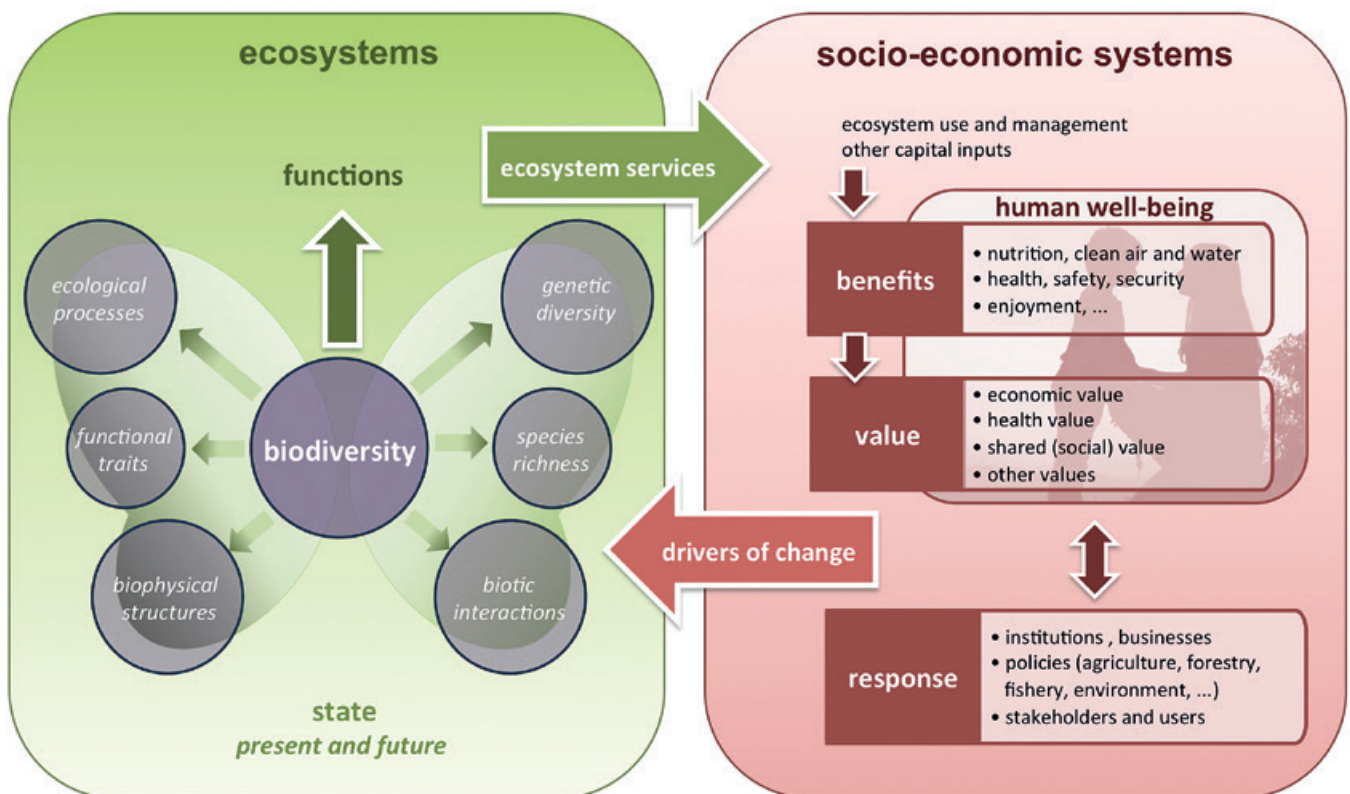

Figure 1. The conceptual framework drawn up by the MAES initiative (Maes *et al.*, 2013a). It links socio-economic systems with ecosystems via the flow of ecosystem services and through the drivers of change that affect ecosystems either as consequence of using the services or as indirect impacts due to human activities in general.

Natural capital

Ecosystem capital

Ecosystems as asset:

Extent, structure and condition of:

Forests, woodlands, rivers, lakes, oceans, coasts, wetlands, grasslands, croplands, heathlands, urban parks, etc.

FRAGILE &

GENERATING

Ecosystems service flows:

- Provisioning services (food, fibre, energy etc.)
- Regulation & maintenance (of climate, river flow, pollination etc.)
- Cultural services (recreation in nature, spiritual use of nature etc.)

DEPLETABLE

Abiotic assets

Solar radiation

NON-DEPLETABLE

Minerals, fossil fuels, ozone layer, gravel, etc.

DEPLETABLE

Abiotic flows

Renewable energy (solar, wind, hydro)

NON-DEPLETABLE

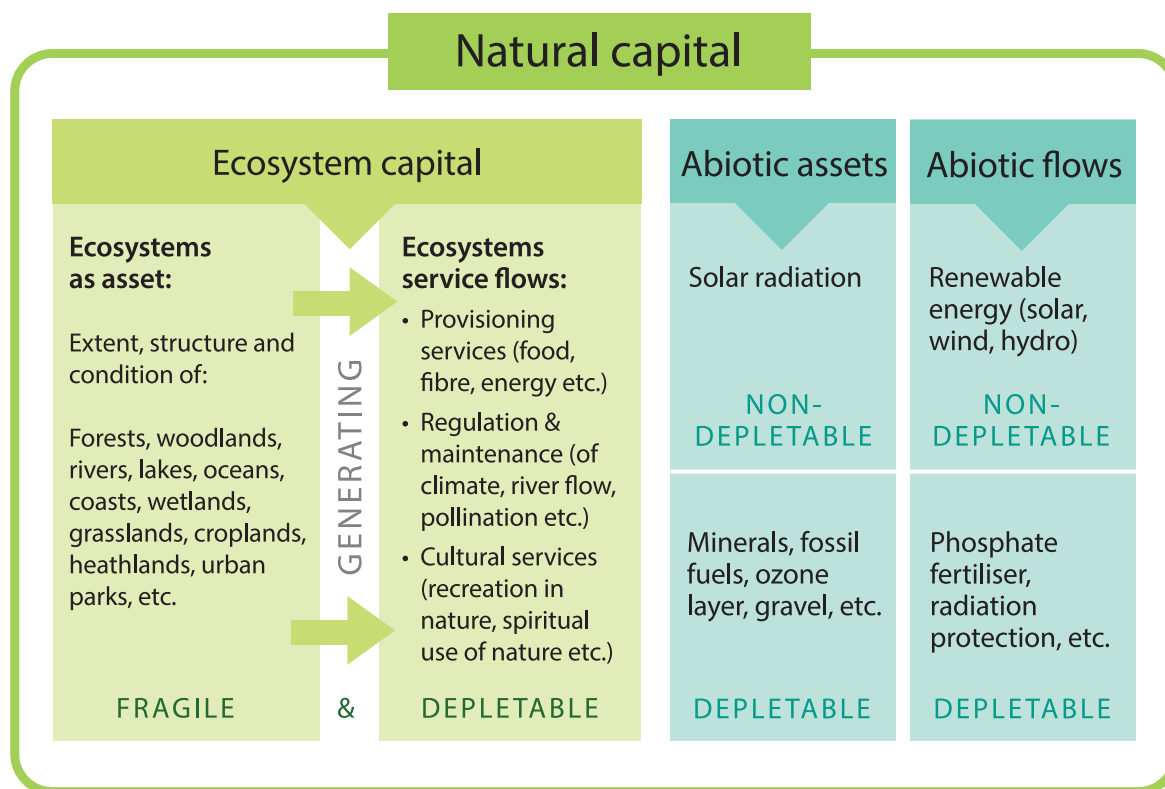Phosphate fertiliser, radiation protection, etc.

DEPLETABLE

Figure 2. Adapted from EEA (2015), illustrating the different components of our natural capital, encompassing both ecosystem stocks and service flows.

assess the state of ecosystems and their services in their national territory by 2014, assess the economic value of such services, and promote the integration of these values into accounting and reporting systems at EU and national level by 2020" (Maes *et al.*, 2014). To this end, the Mapping and Assessment of Ecosystems and their Services (MAES) initiative was set up, and produced a framework for ecosystem assessment to ensure a harmonised approach across the EU (Maes *et al.*, 2013a). This work also contributes to progress towards assessing ecosystem services on a global level, co-ordinated by the Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES) established by the UN in 2012.

MAES uses the Common International Classification of Ecosystem Services (CICES) system for classifying ecosystem services. CICES builds on the MA and TEEB approaches but aims for a system more suitable for accounting. It includes three categories:

i. *Provisioning services.*

ii. *Regulating and maintenance services.*

iii. *Cultural services.*

Provisioning and cultural services are given the same definition as within the MA. The regulating and maintenance service category includes services such as soil formation, originally included as supporting services by the MA, because these are often only indirectly consumed or used (Maes *et al.*, 2013a). Pooling regulating and supporting services in this way tailors the approach to economic accounting, as it helps avoid 'double counting' – when an ecosystem service is wrongly included more than once.

The MAES initiative also makes clear that ecosystem services represent *"the realized flow of services for which there is demand"*. Thus, our 'natural capital' might encompass *stocks* – a forest for example – but the provisioning service itself is the *flow* of harvested timber (see Figure 2 on page 5). In some cases a 'flow' is harder to quantify – enjoyment of a beautiful view, for example – but the basic concept remains the same: an ecosystem service is only an ecosystem service when it is providing a realised benefit to people.

Issues explored in this report

A major criticism of the ecosystem services concept, and a pressing concern highlighted in the MAES reports, is that despite its inclusion in biodiversity policies at national, regional and global levels, protection of ecosystem services may not guarantee protection of biodiversity. Some scholars argue that relying on the ecosystem services approach to halt the biodiversity decline is misguided, as the relationship between biodiversity and ecosystem services is not yet entirely clear (Norgaard, 2010; Faith, 2012; Reyers *et al.*, 2012). In **Chapter 1** we provide an overview of the latest research on the links between biodiversity and ecosystem services and explore the question: *will implementation of the ecosystem services framework also protect biodiversity?*

In order to incorporate the ecosystem services concept into policy and management, decision makers need tools which allow them to assess the supply of services and compare alternative actions. In **Chapter 2** we examine mapping techniques which quantify state of ecosystems and their services and how these change over time and with changing policies. We identify knowledge gaps and solutions to these challenges.

One key goal of the concept is to make explicit the benefits that ecosystems provide. In **Chapter 3** we explore ways to quantify these benefits through valuation. Economic valuation is the most common method and the need to assess the economic value of ecosystem services is stipulated in the EU Biodiversity Strategy; we therefore examine economic valuation techniques, exploring their strengths and weaknesses. We also acknowledge the need to move beyond economic valuation and examine the progress being made towards *integrated valuation,* which can incorporate multiple value systems.

Finally, in **Chapter 4** we highlight a theme that runs through the entire report: the need to consider ecosystem services as part of a wider system. The focus on maximising provisioning services – such as food or timber production – is estimated by the MA to be the single largest driver of biodiversity loss over the last 50 years. The synergies and trade-offs among multiple ecosystem services and the fundamental role of biodiversity, both now and in the future, must be considered.

Aerial photography - The River. ©iStock.com/digitalgenetics

1. The role of biodiversity in ecosystem services

1.1 Introduction

A common criticism of the concept of ecosystem services is that its anthropocentric focus excludes the idea of ecosystems and biodiversity as inherently valuable, beyond human needs (Schröter *et al.*, 2014; Reyers *et al.*, 2012; Deliège & Neuteleers, 2014). Concerns have also been raised regarding the utilitarian nature of the concept, based, as it is, on human benefits (Raymond *et al.*, 2013). Ultimately, it relates only to what ecosystems do for us.

Several arguments counter these criticisms. Researchers point out that implementation of the ecosystem services framework (referred to here as the ecosystem services approach) can be used alongside the idea of the intrinsic value, or indeed even incorporate it to some extent through the inclusion of cultural services such as the spiritual or aesthetic value of a landscape (Schröter *et al.*, 2014; Reyers *et al.*, 2012). Nevertheless, these points lead to one key concern: *if biodiversity is not protected for its own sake, will the ecosystem services approach also protect biodiversity?*

> '*If biodiversity is not protected for its own sake, will the ecosystem services approach also protect biodiversity?*'

The assumption that ecosystem services protection will equate to biodiversity protection is central to the inclusion of the ecosystem services approach in EU policy. As a key part of Target 2, the approach forms part of the strategy to halt biodiversity loss by 2020. However, concern remains over uncertainty regarding the links between biodiversity and ecosystem services (Maes *et al.*, 2014). Critics worry that, if these links are not as clear-cut as previously thought, maximising only those aspects of ecosystems that provide services to humans will not necessarily protect biodiversity, and the drive to make this concept an integral part of many policies will not provide the sustainability intended (Schröter *et al.*, 2014).

To review the evidence we need first to understand what is meant by the term 'biodiversity'. This is widely assumed to mean the number of different species present, but in reality is more complex. The definition used by Convention on Biological Diversity, which is also used within the EU MAES framework, is this: *"The variability among living organisms from all sources including,* inter alia, *terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems"* (Maes *et al.*, 2014, http://www.cbd.int/convention/articles/default.shtml?a=cbd-02).

The definition can be broken down into three main points. Firstly, it focuses on variability, this is crucial because while it includes aspects such as the relative abundances of species, it excludes measures based only on abundance or amount of a single species or group (Mace, Norris & Fitter, 2012). In other words, the total harvest of cabbages may be a measure of an ecosystem service but it is not, in itself, a measure of biodiversity. Secondly, as well as the commonly understood definition of biodiversity as the variation between species (e.g. species richness) it also incorporates diversity within species, including aspects such as genetic diversity. The traditional understanding of species diversity has also been expanded to include the diversity of functional traits – the properties of species that define their ecological role and therefore their impact on ecosystem function and services (Harrison *et al.*, 2014; Lavorel, 2013). Thirdly, it includes the variability of ecosystems themselves, taking into account variation at landscape scales such as between major vegetation types (Mace, Norris & Fitter, 2012). Abiotic or non-living diversity, such as topography, can also interact with biodiversity with important consequences for ecosystem services; however, in this chapter we limit ourselves to discussion of biodiversity alone.

As well as the broad and complex nature of both biodiversity and ecosystem services, the links between them are many and varied. Biodiversity may underpin some ecosystem services but not others; it may provide few improvements to ecosystem services in the short term but aid sustainable, long-term provision. However, despite the complexities, there has now been over 20 years' worth of research and some conclusions can be drawn.

1.2 How does biodiversity affect ecosystem functioning?

To understand how biodiversity affects ecosystem services, we first need to know how it affects ecosystem 'functioning': the stocks and flows of energy and materials and the roles played by primary producers, consumers and decomposers. Research into the links between biodiversity and ecosystem functioning began in the 1980s and an extensive body of research has now been produced. In a seminal review of this diverse evidence, spanning different ecosystems and conditions, Cardinale *et al.* (2012) concluded that: *"There is now unequivocal evidence that biodiversity loss reduces the efficiency by which ecological communities capture biologically essential resources, produce biomass, decompose and recycle biologically essential nutrients."*

The wealth of examples of studies detailing such results encompasses forests (e.g. Cong *et al.*, 2015; Zhang, Chen & Reich, 2012; Piotto, 2008), grasslands (e.g. Finn *et al.*, 2013; Isbell & Wilsey, 2011; Cong *et al.*, 2014; Cardinale *et al.*, 2011), freshwater (e.g. Vaughn, 2010; Cardinale *et al.*, 2011) and marine ecosystems (e.g. Gamfeldt *et al.*, 2014; Cardinale *et al.*, 2011; Worm *et al.*, 2006). Advances have also been made in less well-studied areas, such as soils, showing, for example, that reduced soil biodiversity can reduce nitrogen cycling and plant diversity (Bardgett & van der Putten, 2014; Wagg *et al.*, 2014; de Vries *et al.*, 2013).

Moreover, evidence is mounting to show that the short-term nature of many experiments may underestimate the importance of biodiversity to ecosystem functioning (Cardinale *et al.*, 2012). A large number of studies, for example, have shown the relationship between biodiversity and plant biomass to level off as biodiversity increases (see Figure 3 on page 8). In other words, at high levels of biodiversity the loss of a species has less of an effect than at lower levels. However, the results of Reich *et al.* (2012) suggest that the length of such studies (typically around two years) may not be sufficient to give the whole picture. Using data from two long-term grassland experiments running for at
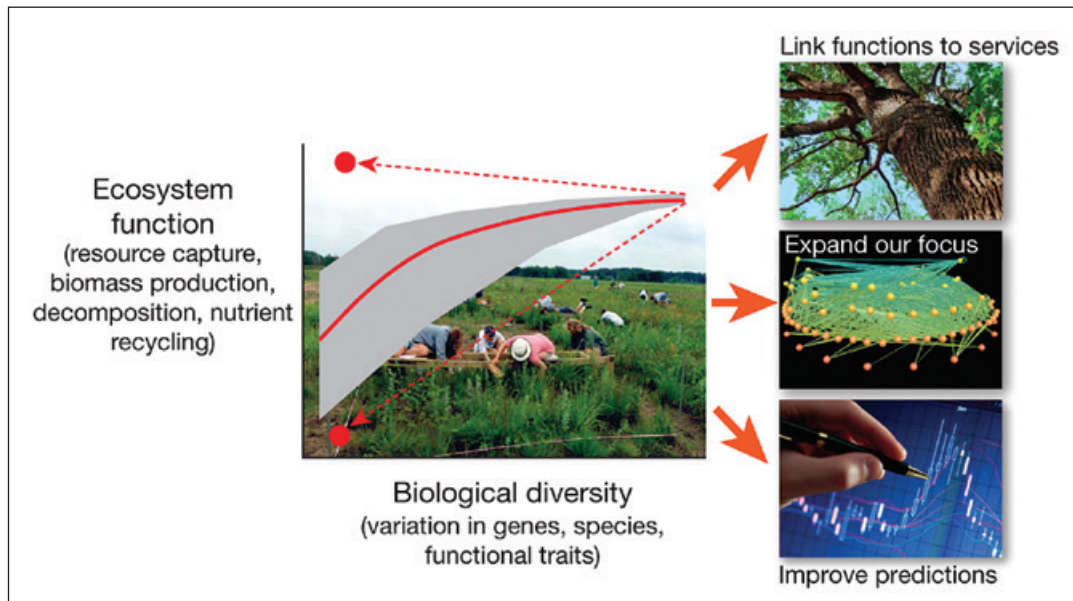
Figure 3. From Cardinale *et al.*, 2012. This figure shows the basic shape of the relationship between biodiversity and ecosystem functions, based on over three decades of research. Although the curve appears to level off at high levels of biodiversity emerging evidence from longer-running experiments suggests that this may not be case if longer timescales are considered (see main text).

least 13 years, Reich *et al.* (2012) demonstrated that, in the short term, the difference in biomass production between plots with medium and high species richness was negligible; however, this difference increased over time, with productivity becoming significantly higher in plots with high biodiversity. Therefore the loss of species from even very biodiverse communities could impair ecosystem functioning.

1.3 How does biodiversity affect ecosystem stability and resilience?

It is important to remember that maximising a single or few ecosystem services in the short term is not the aim of using the ecosystem services concept in policy. The technical report of the MAES working group (Maes *et al.*, 2014) states that research should investigate *"the multifunctionality of ecosystems for sustaining long-term human well-being."* The long-term, stable provision of ecosystem services is therefore the definitive goal. Hence, in examining the links between biodiversity and ecosystem services we must also examine the impacts of biodiversity loss on the stability and resilience of ecosystems.

Stability may be critical as ecosystems come under increasing pressure from myriad anthropogenic drivers, from climate change to invasive alien species. Furthermore, these drivers may have a dual effect: a direct impact on ecosystem services, and an impact on biodiversity, which in turn can affect ecosystem services.

> *'…maximising a single or few ecosystem services in the short term is not the aim of using the ecosystem services concept in policy […] long-term, stable provision of ecosystem services is the definitive goal.'*

There is now good evidence that, as a general rule, increased biodiversity has a stabilising effect on ecosystem functions over time (Thibaut & Connolly, 2013; Cardinale *et al.*, 2012; Jiang & Pu,

2009; Loreau & de Mazancourt, 2013). For instance, one measure of biodiversity, mentioned above, is that of functional diversity. This is a measure of the diversity of ecological roles that are needed for an ecosystem to function. If a number of species appear to perform the same role there is presumed to be 'functional redundancy': in other words it is assumed, based on current knowledge, that not all species are needed for the ecosystem to function (Mori, Furukawa & Sasaki, 2013). However, in the face of global change, having a number of different species performing similar roles may be vital. Stability is likely to be higher if more than one species perform the same function because a decline in one species may be compensated for by stable or increasing numbers of another, especially if they respond differently to disturbances and environmental change (Loreau & de Mazancourt, 2013; Mori, Furukawa & Sasaki, 2013; Winfree, 2013).

Diversity *within* a species can also have a stabilising effect. High genetic diversity, for instance, can make species more resilient against stressors and quicker to adapt to environmental change. A review by Hajjar, Jarvis & Gemmill-Herren (2008) concludes that increased genetic diversity in crop species generally improves resistance to pests and diseases and enhances pollination. Hughes & Stachowicz (2004) demonstrate that higher genetic diversity within the seagrass species *Zostera marina* enhances community resistance to disturbance by grazing geese.

Biodiversity in the form of life history traits, such as the age structure of a population or the timing of migration, may also be important to stability. Schindler *et al.* (2010) used a 50-year dataset on sockeye salmon (*Oncorhynchus nerka*) in Alaska's Bristol Bay to examine such effects. Sockeye salmon in the area form several hundred populations which vary widely in terms of life history. The researchers calculated that if the Bay's population had been a single, homogenous entity, the variability of the salmon catch would be twice as high. This increased variability would have led to 10 times more fisheries closures, as salmon numbers would have undergone more extreme peaks and troughs, rather than providing a steady supply.

The specific way in which anthropogenic stressors affect biodiversity could also have important implications for the resilience of ecosystem functioning and services. For example, the order in which species are lost is likely to make a difference. Bunker *et al.* (2005) simulated different orders of species extinction (based on different types of extinction risk, such as small populations, or species most likely to be harvested by loggers) in a 50-hectare tropical forest. They found that the models predicted that carbon stored in aboveground biomass would vary by more than 600%, depending on order of extinction. There is now also good evidence that the reduction of biodiversity across several trophic levels (how far up the food web a species is) is likely to have greater effects on ecosystem functioning than biodiversity loss within trophic levels (Cardinale *et al.*, 2012; Winfree, 2013).

The importance of the order of species loss also raises questions as to how best to prioritise funding for biodiversity and ecosystem services protection. Historic approaches have focused significantly on charismatic species, sometimes justified by their role as 'flagship' species dependent on the integrity and connectivity of whole networks of ecosystems. Yet it is the vast menagerie of poorly understood microbes that drive most nutrient cycling, for example.
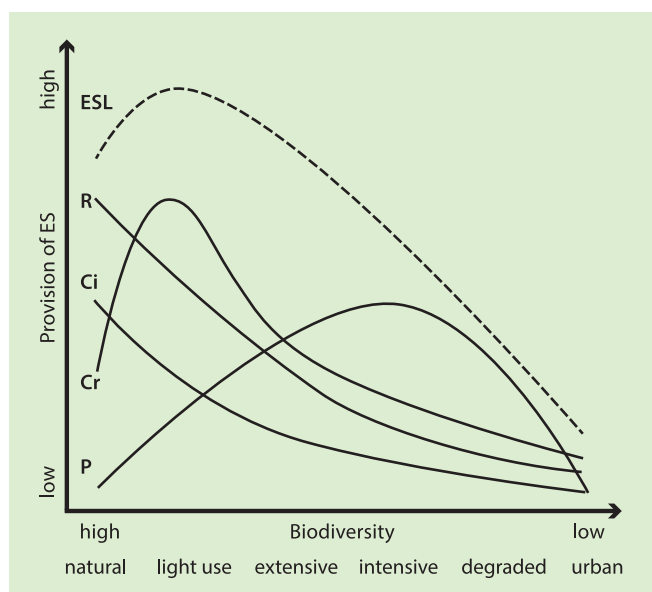

Figure 4. Adapted from Braat & ten Brink (2008). R: sum of regulating services; P: sum of provisioning services; Cr: sum of cultural-recreation value; Ci: sum of cultural-information value (including aspects such as cultural heritage, education, etc.); ESL: sum of all the ecosystem services.

1.4 How does biodiversity affect ecosystem services?

As we have seen above, there is now a firm evidence base demonstrating the importance of biodiversity to ecosystem functioning. However, there is less research into whether biodiversity has the same pivotal role for ecosystem services, and hence whether protection of ecosystem services will protect biodiversity, and vice versa (Harrison *et al.*, 2014).

Figure 4 shows a simplified diagram of how changes in biodiversity are predicted to affect four types of ecosystem services: regulating, cultural-information (including aspects such as cultural heritage, education, etc.), cultural-recreational and provisioning. (Cimon-

Morin, Darveau & Poulin, 2013; de Groot *et al.*, 2010; Braat & ten Brink, 2008). Provisioning services, for example, peak at relatively low levels of biodiversity, and this makes intuitive sense: intensive agriculture provides large amounts of provisions in the form of food, but has severe impacts on biodiversity (MA, 2005).

> '...there is now a firm evidence base demonstrating the importance of biodiversity to ecosystem functioning. However, there is less research into whether biodiversity has the same pivotal role for ecosystem services.'

Maes *et al.* (2012) mapped four provisioning services, five regulating services and one cultural service across Europe, and found that these were positively correlated with biodiversity, although they note that this relationship was affected by trade-offs, in particular between the provisioning service of crop production and regulating services. However, this study examined correlations; to actually determine whether biodiversity has a pivotal role in ecosystem services supply, experimental evidence is needed.

Balvanera *et al.* (2014) examine whether biodiversity, measured as species richness, drives ecosystem services supply for three provisioning services: forage, timber, fisheries; and three regulating services: climate regulation, regulation of agricultural pests and water quality. They caution that while a positive link between biodiversity and ecosystem functioning is now strongly supported there is less evidence of a clear relationship between biodiversity and ecosystem services.

For example, although the fact that higher species richness drives higher biomass in grasslands is well-established (Cardinale *et al.*, 2011), this may not equate to increased forage provision. There have been few studies documenting whether the extra biomass produced contains the nutrients needed for livestock, for instance (Balvanera *et al.*, 2014). Links between biodiversity and timber provision were stronger, with studies showing that increased species richness increased biomass production, regardless of whether a forest stand was natural or plantation (Piotto, 2008; Zhang, Chen & Reich, 2012).

For the regulating service water purification, Balvanera *et al.* (2014) summarised 59 experiments, showing that in 86% of studies, spread across terrestrial, freshwater and marine ecosystems, increased species richness reduced nitrogen concentrations in water or soil. Moreover, species richness was manipulated by the researchers in these studies; therefore they did not conflate correlation with causation. For example, Cardinale (2011) set up 150 experimental streams that mimicked natural conditions and found increasing the number of algal species lead to increased nitrogen uptake. Different species dominated in different habitat niches, allowing biodiversity and nitrogen uptake to remain high.

The key role of biodiversity for regulating services has been verified by other research (Harrison *et al.*, 2014; Mace, Norris & Fitter, 2012, see Figure 4). For example, experiments have shown that bioremediation of contaminated groundwater and marine sediments is faster and more effective when bacterial biodiversity is higher (Dell'Anno *et al.*, 2012;

Marzorati *et al.*, 2010). Pollination services have also been shown to improve as pollinator diversity increases (Winfree, 2013; Brittain, Kremen & Klein, 2013; Garibaldi *et al.*, 2013). The significance of protecting regulating services and the biodiversity that underpins them should not be underestimated, as many other ecosystem services are dependent upon them (Harrison *et al.*, 2014; Maes *et al.*, 2012b).

Some scholars have raised the question of whether biodiversity might ever have negative effects on ecosystem services. Harrison *et al.* (2014), for example, say that freshwater provision can be negatively affected by greater plant coverage, as it increases water retention. However, this effect is linked to the total area of vegetation and age and size of the plant species, rather than any measure of biodiversity. Furthermore, examining a single service in this way, without considering the multi-functionality of the entire system, can lead to limited conclusions that do not consider the overall effect on all ecosystem services and human well-being. For example, slopes cleared of vegetation may maximise freshwater production in the short term but are likely to have damaging effects on soil erosion, water quality and recreation, etc.

The importance of systemic thinking and multi-functionality is not a side issue when it comes to considering the links between biodiversity and ecosystem services. In fact, the true significance of biodiversity may only be revealed when the whole system, across the full spectrum of ecosystem services, including different locations and across many years, is considered. Indeed, evidence is now mounting to show that higher biodiversity is needed to maintain multiple ecosystem services in the long term and under environmental change (Gamfeldt *et al.*, 2013; Balvanera *et al.*, 2014; Cardinale *et al.*, 2012; Isbell *et al.*, 2011; Naeem, Duffy & Zavaleta, 2012).

For instance, Isbell *et al.* (2011) conducted a meta-analysis across grassland biodiversity experiments and concluded that different species were important for different functions at different times, places and under different environmental change scenarios. Biodiversity loss in one place may create an 'ecosystem services debt' in another, for example (see Box 1). Overall, studies which have considered the direct influence of biodiversity for only one ecosystem service, only over a short time period, or without any influence of global change, are likely to underestimate its importance.

View across Westleton Heath, Suffolk, UK. ©iStock.com/coastalrunner

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Box 1

The ecosystems services debt

When assessing the flow of ecosystem services, and how they might change in the future, scientists generally assume that any habitat that remains intact will continue to provide its normal, full range of services (Isbell *et al.*, 2014). However, habitat fragmentation can lead to an 'extinction debt' – a delay between reduction in habitat and extinction of species in the remaining fragments (Kuussaari *et al.*, 2009). For example, an individual of a long-lived tree species may persist in a habitat fragment for a long time, but if there are not enough individuals to enable reproduction the species will become extinct. This could, in turn, lead to an 'ecosystem services debt'. Isbell *et al.* (2014) developed an approach for assessing this phenomenon for the ecosystem service carbon storage. They found that between 2 000 and 21 000 megatonnes of carbon could be gradually released as a result of plant species loss due to habitat destruction in nearby areas. The wide range of the estimate stems from uncertainties surrounding how many plants will be lost, whether it will also affect soil carbon and the effects on ecosystem functioning. Nevertheless, the authors say the results show that ecosystem services debts could be globally substantial and should be accounted for when quantifying the effects of habitat destruction.

1.5 Knowledge gaps

Despite evidence of the importance of biodiversity to sustained provision of many ecosystem services, knowledge gaps remain. Specifically, more long-term, large-scale interdisciplinary research is needed to answer these central questions:

i. How does biodiversity mediate synergies and trade-offs among multiple ecosystem services?

ii. Is biodiversity important to the long-term provision of ecosystem services?

iii. How will the non-random order of species loss under global change affect ecosystem services?

iv. What are the simultaneous effects of different components of biodiversity?

Given the emerging message that focusing on a single ecosystem service may underestimate the importance of biodiversity, more studies that consider the interdependencies of multiple ecosystem services and biodiversity are urgently needed (Balvanera *et al.*, 2014; Harrison *et al.*, 2014). For example, regulating services, which often rely heavily on biodiversity, can be vital in sustaining other ecosystem services. A good example of this is the links between soil formation and nutrient cycling – both regulating/maintenance services with close links to biodiversity – and crop production, a provisioning service which may

appear much less dependent on biodiversity but clearly requires healthy soils and available nutrients. The complexity of these relationships makes research difficult, but not impossible, see Box 2 for a suggested framework of such studies.

Box 2

In assessing the necessary next steps into the links between ecosystem services and biodiversity, Balvanera *et al.* (2014) make a number of recommendations for future studies.

- Meta-analyses can provide insight into the current state of the research and help identify knowledge gaps. Using existing data to create models of the systems under scrutiny can also help develop key questions to be answered.

- A network of sites spanning different ecological and social conditions should be used. Alternative management approaches should be tested, with biodiversity and ecosystem services continually monitored.

- Multiple ecosystem services should be considered, and the synergies and trade-offs between them, as well as the trade-offs between long- and short-term ecosystem services provision.

- Stakeholders should also be consulted, to ensure that studies examine issues that are relevant to them.

While the authors recognise the practical challenges of creating such studies, they point out that examples of this approach do exist (e.g. Garibaldi *et al.,* 2013).

In addition, this kind of monitoring and investigation should not be limited to academic experiments. Projects and initiatives based on an ecosystem service approach should monitor whether their actions also protect biodiversity (Schröter *et al.*, 2014). Conversely, conservation and restoration schemes focused on biodiversity should make an effort to quantify how they affect ecosystem services.

1.6 Conclusions

In summary, decades of research have shown that biodiversity plays a vital role in ecosystem functioning, and that processes such as capturing essential resources, producing biomass and recycling nutrients, are all impaired as biodiversity declines (Cardinale *et al.*, 2012). Furthermore, biodiversity not only underpins ecosystem functioning, it also enables these processes to be resilient in the face of global change (Loreau & de Mazancourt, 2013).

While research into the links between biodiversity and ecosystem services is less well-developed, it is nonetheless beginning to deliver clear messages. One key emerging trend is that a significant part of the value of biodiversity in underpinning ecosystem services is likely to lie in the long-term, stable provision of multiple services in the face of global change (Isbell *et al.*, 2011; Gamfeldt *et al.*, 2013; Balvanera *et al.*, 2014). The exact relationship between biodiversity and each individual ecosystem service varies, but if the system can be examined as a whole, accounting for trade-offs and synergies between

all services, the fundamental nature of biodiversity will be revealed. This demonstrates that we must take a fully systemic approach to the contribution of biodiversity to societal wellbeing and that wilfully or inadvertently favouring some services over others risks loss of system integrity, functioning and resilience.

The evidence presented here suggests that the answer to the question 'Is biodiversity important for ecosystem services provision overall?' is an unequivocal 'Yes'. However, as the EU increasingly uses the ecosystem services approach as a way to halt the decline in biodiversity and a cornerstone of sustainability in general, a perhaps more relevant question for policymakers and conservationists is the one asked at the beginning of this chapter: 'Will use of the ecosystem services approach protect biodiversity?'

There is much research to be done into the many uncertainties surrounding this question (see Section 1.5). However, the evidence to date suggests that truly sustaining the long-term flow of all ecosystem services will require high levels of biodiversity. Thus, a 'no regrets' policy position would be to recognise biodiversity as non-substitutable capital underwriting human well-being, rather than allow its continued degradation as we continue to study the many ways in which biodiversity contributes to a viable future for humanity.

However, we must prepare for the possibility that, even if multi-functionality, stability, resilience and interdependencies are all accounted for, some biodiversity is unnecessary for ecosystem services provision. If this is the case, will the use of the ecosystem services approach mean that biodiversity is lost? Not necessarily. Many scholars argue that the ecosystem services approach does not need to replace policies designed to protect biodiversity *per se* but can work alongside them highly effectively (Reyers *et al.*, 2012; Faith, 2012).

> *'The answer to the central question — will use of the ecosystem services approach protect biodiversity? — is likely to be a qualified yes.'*

Indeed, there may well be important synergies between these two approaches. For instance, biodiversity projects often focus on providing relatively unspoilt reserves that can form important refuges, such as the Natura 2000 network; complementing this, the ecosystem services approach works more widely, integrating the importance of biodiversity into urban planning, for example (Reyers *et al.*, 2012).

The EU Biodiversity Strategy makes the dual importance of ecosystem services and the intrinsic value of biodiversity explicit in its vision for 2050: *"By 2050 European Union biodiversity and the ecosystem services it provides – its natural capital – [will be] protected, valued and appropriately restored for biodiversity's intrinsic value and for their essential contribution to human wellbeing and economic prosperity, and so that catastrophic changes caused by the loss of biodiversity are avoided."*

In summary, the answer to the central question – will use of the ecosystem services approach protect biodiversity? – is likely to be a qualified yes. As long as the approach is implemented via policies based on sound evidence, and in conjunction with strategies that recognise the intrinsic value of biodiversity, it has the potential to be a powerful instrument in the struggle to halt biodiversity decline.

2.   Mapping and assessing ecosystem services

2.1 Introduction: Why map ecosystem services and how to go about it?

Implementing the ecosystem services approach effectively will require decision makers and other stakeholders to understand the trade-offs and synergies between multiple ecosystem services and biodiversity. Maps of ecosystem services are the first and most important tool in this process. Mapping can provide information for a number of crucial issues; for instance, such exercises can be used to examine:

i.   How optimising ecosystem services can also benefit biodiversity, and *vice versa* (Willemen *et al.*, 2013)

ii. The trends in provision of ecosystem services, and how different drivers affect them over time (Malinga *et al.*, 2015)

iii. The synergies and trade-offs between multiple ecosystem services (Queiroz *et al.*, 2015; Bennett, Peterson & Gordon, 2009)

iv. The costs and benefits of maximising ecosystem services (Schägner *et al.*, 2013)

v.   How supply and demand varies spatially (Schulp, Lautenbach & Verburg, 2014)

This can help answer important questions such as where to invest to ensure the stable delivery of multiple services and protection of biodiversity. In addition, such maps can provide a valuable stakeholder communication tool, illustrating the interplay among different ecosystem services at a range of spatial scales (Hauck *et al.*, 2013).

The process of mapping and assessing ecosystems and their services begins with mapping ecosystems themselves. Large-scale land cover maps produced using high-resolution satellite imagery, available from programmes such as CORINE Land Cover[1] , have proved valuable for this. They can be linked to habitats, and hence ecosystems, using the European Nature Information System (EUNIS) classification and data on elevation, or geological conditions (Maes *et al.*, 2014). A full map of European ecosystems has now been completed by the European Environment Agency (EEA)[2] .

The next step is to assess ecosystem condition, defined as the physical, chemical and biological quality of an ecosystem. Ecosystem condition is a vital part of the assessment, because it dictates the capacity of an ecosystem to yield services (Maes *et al.*, 2014; EEA, 2015). Drivers and pressures, such as the intensification of agriculture, water pollution or climate change, can all reduce ecosystem condition and impair the delivery of ecosystem services (EEA, 2015).

The second MAES report, *Indicators for ecosystem assessments under Action 5 of the EU Biodiversity Strategy to 2020*, advocates combining assessments of ecosystem condition and services to provide an integrated ecosystem assessment (see Figure 5 on page 13).

Methods of mapping and assessing ecosystem services supply vary between services. For some, such as food or timber provision, primary data on actual outputs may be already available, giving ecosystem services maps with good levels of accuracy (Maes *et al.*, 2012a). However, many ecosystem services are not monitored in this way and mapping is often fundamentally based on large-scale land cover/land use maps, though increased layers of complexity can be added in a tiered approach:

Tier 1: In the simplest form of mapping, experts provide a score of ecosystem services supply for each type of land cover and these scores can then be used to directly map services supply from the land cover map itself (Burkhard *et al.*, 2009, see Figure 6 on page 14). For example, experts use their knowledge to estimate the water purification provided by a given area of forest and the area of forest is then used as a proxy or 'indicator' for the supply of the water purification. Using this approach may be suitable for where the ecosystem service is closely related to land use, such as food provision, but may be less accurate in cases where more detailed, smaller-scale maps are needed, or where there is no clear relationship between land cover and the ecosystem service of interest (Maes *et al.*, 2012a; Maes, Paracchini & Zulian, 2011).

Tier 2: Tier 2 maps build on the tier 1 approach by incorporating extra data to add detail and accuracy. For example, primary data collected in one area can be 'upscaled': linked to land cover data and used over larger scales. Data can also be 'downscaled' when, for example, national timber statistics are disaggregated and used to map provision of this service over more local scales.

Tier 3: The third tier adds another level of detail by incorporating 'process-based models'. These models account for the underlying processes, both biological and physical, that affect the supply of an ecosystem service. For example, soil type, mix of plant species and topology might all affect water purification. Process-based models can also be useful in predicting how ecosystem services supply will be affected by changes in drivers or pressures in the future. The tier 3 approach combines all forms of information to ensure the highest possible accuracy: process-based models can be validated using data on actual ecosystem services supply, or analysis based on primary data can be used in conjunction with process-based models (Schulp *et al.*, 2014; Schägner *et al.*, 2013, see Figure 2). Box 3 provides an example of a study which takes a tier 3 approach, incorporating numerous different types of data and several models.

1. http://www.eea.europa.eu/publications/COR0-landcover
2. http://projects.eionet.europa.eu/eea-ecosystem-assessments/library/draft-ecosystem-map-europe

(1)
Map ecosystems

Urban
Cropland
Grassland
Woodland and forest
Heathland and shrub
Sparsely vegetated land
Wetlands
Rivers and lakes
Marine inlets and transitional waters
Coastal
Shelf
Open ocean

Land use land cover data, e.g.
Corine Land Cover
Copernicus high resolution data
Elevation data
Seabed maps
National datasets

Models for spatially delineating wetlands or natural, unmanaged systems

(2)
Assess the condition of ecosystems

Indicators	Data
Conservation status of habitats and species	Art.17 assessment
Ecological status of water bodies	WFD assessment
Environmental status of seas	MSFD assessment
Ecosystem status and biodiversity	data including air pollutant concentration, habitat connectivity, land use change, soil degradation, ...

(3)
Assess te ecosystem services delivered by ecosystems

Indicators	Data and models
Supply indicators: Indicators of stock and flow of ecosystem functions and ecosystem services	Different sources of environmental data and models
Demand indicators: Indicators for the human demand for ecosystem services	Different socio-economic statistics

(4)
Integrated ecosystem assessment:
How does condition relate to services provision?
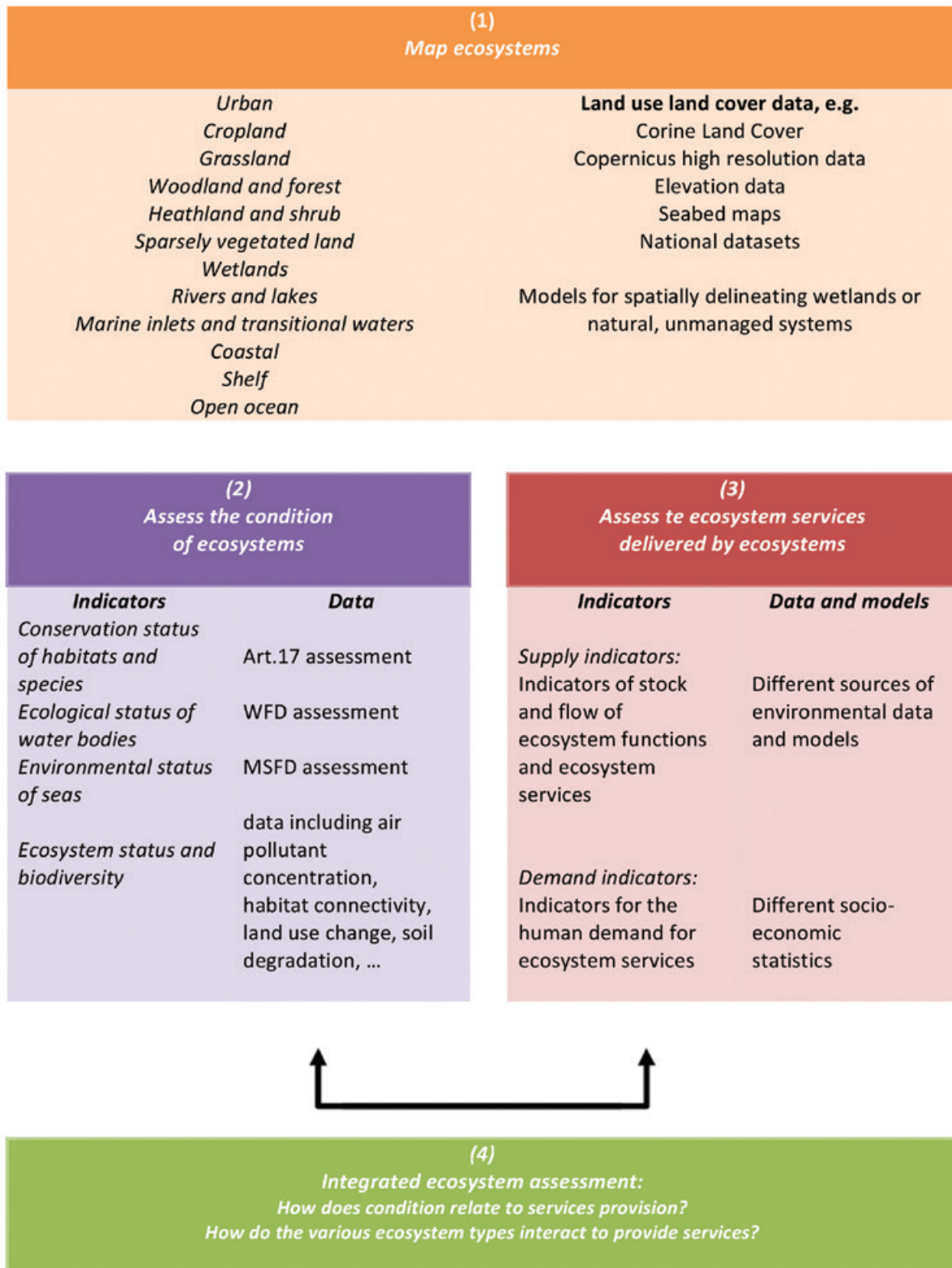**How do the various ecosystem types interact to provide services?**

Figure 5. From Maes *et al.* (2014). This figure illustrated the framework for integrated ecosystem assessment drawn up by the MAES initiative.
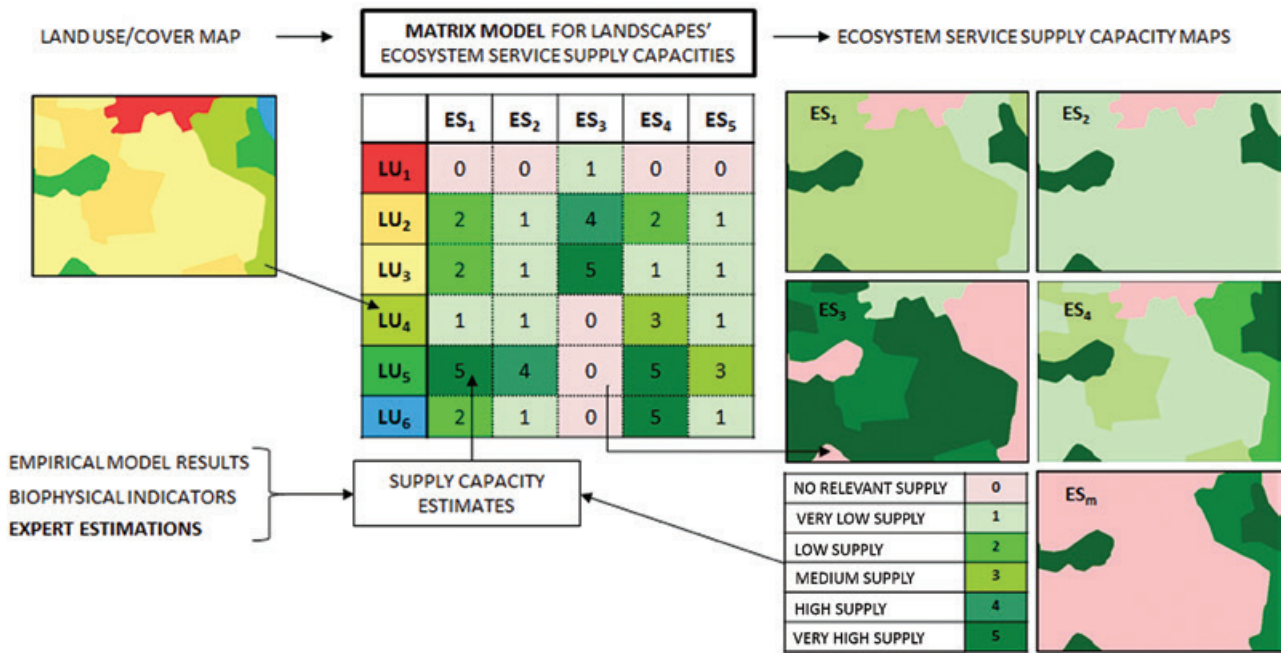
Figure 6. From Jacobs *et al.* (2015). This figure illustrates the use of land cover maps to provide maps of ecosystem services supply. Supply capacity estimates can follow a tier 1, 2 or 3 approach (see main text) adding increasing layers of complexity.

The second MAES report (Maes *et al.*, 2014) collates a large number of indicators that can be used to map and assess ecosystem services at the national level, and these have now been used in six detailed pilot studies in different EU Member States. For example, in the pilot study to map ecosystem services provided by forests, over a hundred indicators are listed that could be used to assess and map provisioning, regulating/maintenance and cultural services.

These indicators are categorised under the CICES method (see Figure 7). This hierarchical system is particularly useful to group services together where only higher-level indicators are available. It also ensures that indicators that have been developed for national scales, for example, can still be used for international assessments (Maes *et al.*, 2014).
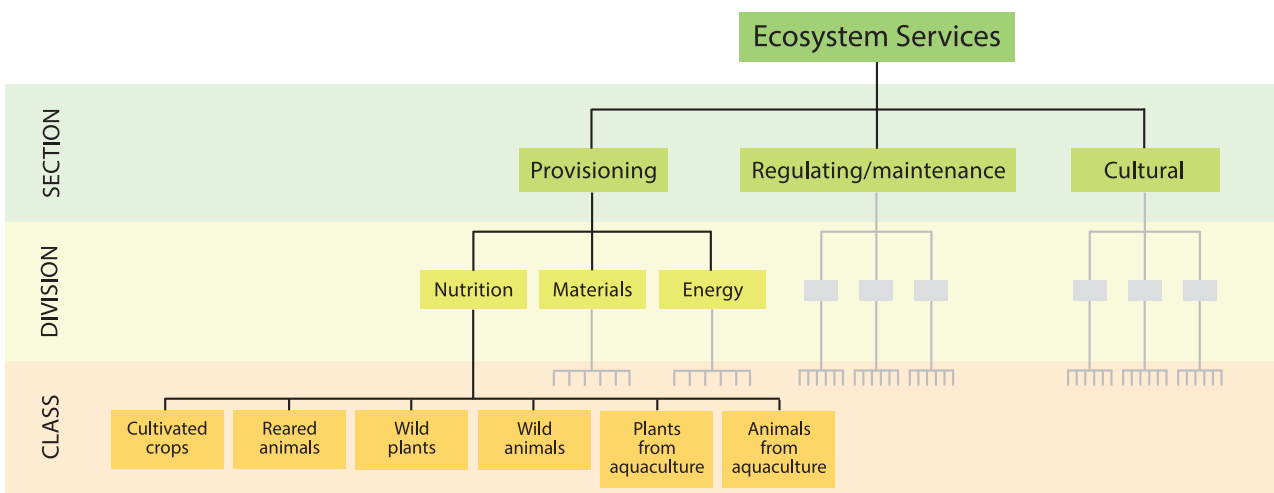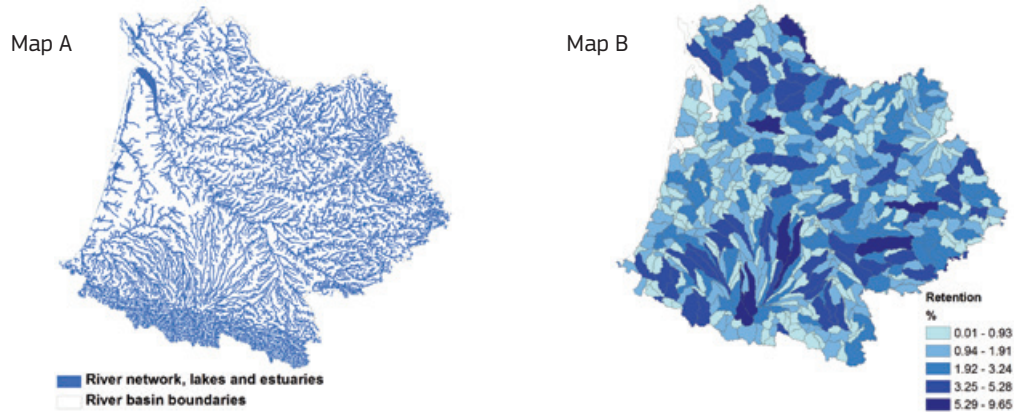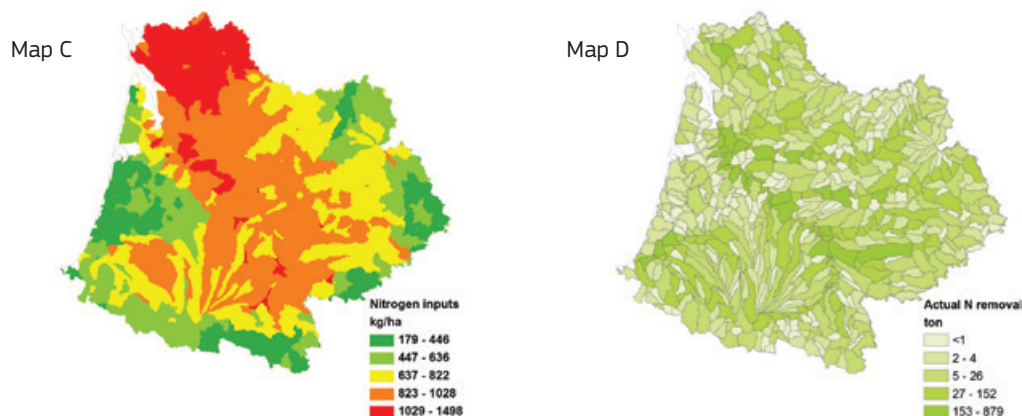

Figure 7. Illustrating the CICES hierarchical system of classification (author's own diagram).

Box 3

Here we describe a case study of the steps involved in mapping nitrogen removal, an indicator for the water purification, in the French river basin district of the Adour and Garonne, as detailed by Maes *et al.* (2012).

Map A

Map B

Retention
%
- 0.01 - 0.93
- 0.94 - 1.91
- 1.92 - 3.24
- 3.25 - 5.28
- 5.29 - 9.65

■ River network, lakes and estuaries
□ River basin boundaries

Map A shows the physical structure of the rivers and streams in the basin, an important initial step, as this partly dictates how water flows through the system. A model then calculates the retention of nitrogen in soils and surface water, allowing researchers to map the capacity of ecosystems to provide this service (Map B).

Map C

Map D

Nitrogen inputs
kg/ha
- 179 - 446
- 447 - 636
- 637 - 822
- 823 - 1028
- 1029 - 1498

Actual N removal
ton
- <1
- 2 - 4
- 5 - 26
- 27 - 152
- 153 - 879

Map C shows the nitrogen input into the area. Map D is dependent on a model which uses information from both Map B and Map C to show the realised service: the actual amount of nitrogen removed from the water. These maps can now be used to assess the impacts of different policy measures. The benefits of restoration of wetlands, for example, can be evaluated. Importantly, for this exercise the researchers also mapped sustainable nitrogen removal, in other words, the amount of nitrogen that can be removed while ensuring that the ecosystem is not harmed and the ecosystem service can continue to be delivered in the long term (Map E).

Map E

Sustainable N.
removal
ton
- <1
- 2 - 4
- 5 - 26
- 27 - 152
- 153 - 879
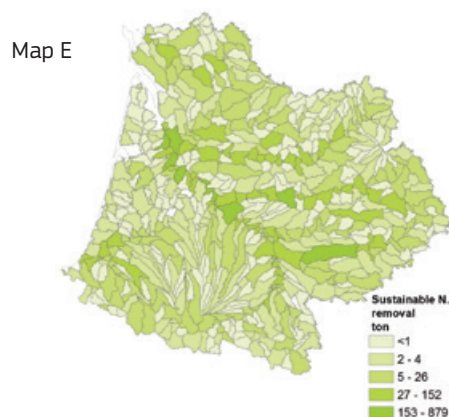
To do this, they set a maximum concentration of 1 mg of nitrogen per litre, on the assumption that environmental harm does not occur below this threshold. This map is crucial, since it integrates data on ecosystem condition with those on ecosystem services. Comparison between Map D and Map E, for example, can provide information on where the ecosystem is being used to provide services unsustainably.

2.2 Challenges for mapping ecosystem services

While great progress has been made in the EU towards mapping and assessing ecosystems and their services, challenges and knowledge gaps remain (Egoh *et al.*, 2012; Maes *et al.*, 2012a; Martínez-Harms & Balvanera, 2012).

2.2.1 Data availability

Data availability is a key issue and it can vary greatly between ecosystem services. As a result of the lack of primary data, many mapping exercises rely on proxies to convert land cover maps to ecosystem services maps (as discussed above). This creates a bias towards mapping those ecosystem services that can be linked to land cover maps in a straightforward manner, such as regulating services like carbon storage. In a review of studies mapping ecosystem services Malinga *et al.* (2015) found that regulating services were mapped most often, in 46% of studies, a trend which echoes findings in other reviews (Crossman *et al.*, 2013; Martínez-Harms & Balvanera, 2012; Egoh *et al.*, 2012).

> *'More data is needed, particularly for services such as genetic resources, and many cultural services.'*

The variability in the amount and quality of the data available for mapping different ecosystem services is illustrated in the second MAES report: harmonised, spatially explicit data at European scale was available for only 15% of the indicators suggested for forests, but for agro-ecosystems this was 27%.

More data is needed, particularly for services such as genetic resources, and many cultural services. Marine ecosystems are also under-studied in comparison to terrestrial ones, meaning that knowledge of functional relationships, which have been widely used to map terrestrial services, is poor (Guerry *et al.*, 2012). There is concern that if suitable data cannot be found for these ecosystem services they will be neglected in policy decisions as a result (Maes *et al.*, 2012a).

2.2.2 Mapping multiple services

The importance of considering multiple services simultaneously should not be underestimated (see Chapter 4). Evidence shows that maximising one ecosystem service without considering the whole system can and has had damaging consequences for both other services and biodiversity (Bennett, Peterson & Gordon, 2009; MA, 2005; Everard & McInnes, 2013).

Despite the primary significance of this issue, many studies map only a single or few ecosystem services (Crossman *et al.*, 2013; Maes *et al.*, 2014; Martínez-Harms & Balvanera, 2012). For instance, a review by Crossman *et al.* (2013) showed that out of 113 mapping studies 32% mapped only one ecosystem service.

Those studies that have examined multiple ecosystem services have found complex trade-offs and synergies, meaning that no single service can be viewed as a proxy for others and therefore management to maximise one will not suffice to protect the multi-functionality of the system (Bennett, Peterson & Gordon, 2009; Queiroz *et al.*,

2015). Conversely, if research can provide clear information about the relationships between ecosystem services, management could be designed to enhance multiple services.

2.2.3 Mapping both supply and demand

Overall, the supply of ecosystem services has been mapped much more frequently than demand (Crossman *et al.*, 2013; Bagstad *et al.*, 2014; Burkhard *et al.*, 2014). However, as defined above, ecosystem services are the benefits that people obtain from ecosystems; therefore, in the strictest sense, an ecosystem service which goes unused by humans is not an ecosystem service.

Distinguishing between supply and demand, and mapping both, is vital, as it allows decision makers to ensure that demand for ecosystem services, which is rising globally, does not exceed the capacity of ecosystems to supply them (Crossman *et al.*, 2013). Furthermore, it is important that demand is considered separately in mapping studies because it may vary over time and space in ways that are entirely independent of supply (Burkhard *et al.*, 2014). For example, a remote forest may produce a large amount of wild berries but if people are unable to access the area there is no flow of services. The scales of demand may also vary between services. Demand for regulating services often has to be met locally or regionally. Flood regulation, for example, occurs close to the point of supply, such as upriver from a village. However, demand for many provisioning services can be vast distances from supply. Fruit grown in New Zealand may be consumed in Europe, for example.

> *'Distinguishing between supply and demand [...] allows decision makers to ensure that demand for ecosystem services, which is rising globally, does not exceed the capacity of ecosystems to supply them.'*

2.2.4 Uncertainty

It is vital to recognise that the complexity of the processes underpinning ecosystem services, the lack of data for many services and the subsequent reliance on model-based proxies means that these maps are associated with inherent uncertainties (Schulp *et al.*, 2014; Eigenbrod *et al.*, 2010; Egoh *et al.*, 2012; Jacobs *et al.*, 2015). However, many studies do not provide quantitative estimates of the magnitude of errors. Seppelt *et al.* (2011) reviewed 153 ecosystem services mapping studies and found that 45–80% of studies did not give sufficient information on uncertainty or validation. Examining 79 studies used to map ecosystem services supply in terms of economic value, Schägner *et al.* (2013) found that over a third did not address accuracy at all.

Eigenbrod *et al.* (2010) provided quantitative evidence of ecosystem services mapping errors, showing that land cover based proxies show significant errors when compared to primary data, especially at smaller spatial scales. Furthermore, this was shown to have implications not only for the accurate mapping of a single service but for understanding of the links among services: correlations between services varied depending on whether primary or proxy data were used (Eigenbrod *et al.*, 2010).

The intrinsic uncertainty in ecosystem services mapping is also illustrated by Schulp *et al.* (2014) who compared four well-known, published sets of ecosystem services maps of Europe for four different services: climate regulation, recreation potential, soil erosion protection and flood regulation. To compare the maps, the researchers used a Map Comparison Statistic (MCS). This was based on a scale from zero to one, where two identical maps score zero and two completely opposite maps score one. The results showed that the four of maps of climate regulation and those of recreation potential were in broad agreement within their sets, with MCS values of 0.28 or less. Pollination maps showed intermediate agreement, with MCS values ranging from 0.20–0.49. Flood regulation and erosion protection maps showed the lowest levels of agreement, with MCS values of 0.17–0.53 for flood regulation and 0.26– 0.64 for erosion protection. These results do not reflect the accuracy of the maps but they do highlight the considerable variation between methods and uncertainty in ecosystem services mapping.

Jacobs *et al.* (2015) discuss measures that can be used to improve the situation. They recommend that ecosystem services mapping projects, especially those that incorporate expert judgment, should follow the method of 'confidence reporting' used by the Intergovernmental Panel on Climate Change (IPCC). The IPCC method generates measures of confidence based both on scientific evidence itself and degree of agreement between researchers. Furthermore, models should be checked for reliability and validated using both primary data and expert opinion from different sources. Bayesian statistical models are likely to prove an important tool, as they can incorporate diverse data types (scores, primary data, datasets with missing data) and quantify accumulated uncertainties (Jacobs *et al.*, 2015).

Uncertainty in ecosystem services mapping does not, by any means, invalidate the approach. As discussed above, ecosystem services maps can prove a valuable tool for policymakers and managers for a variety of different purposes and this remains true even if some uncertainty exists. However, recognition of uncertainty, and quantification of the levels of uncertainty, is vital for map users. Decision makers need to know whether the maps they are using provide precise, site-specific information or only general, large-scale trends.

2.3 Practical use of ecosystem services maps

A key part of the mapping process should be to take into account the information needs and requirements of decision makers and managers (Albert *et al.*, 2014a). Hauck *et al.* (2013) conducted interviews and focus groups at regional, national and EU levels, including representatives of the EC General Directorates Environment and Agriculture and Rural Development as well as national ministries, regional planners, farmers, researchers and others. They found that respondents felt that the ecosystem services concept was useful for decision support, a result echoed in other studies (Albert *et al.*, 2014b; Mascarenhas *et al.*, 2014). Despite this, inclusion of the concept in Europe is as yet mainly implicit (Hauck *et al.*, 2013; Albert *et al.*, 2014a; Mascarenhas *et al.*, 2014).

Decision makers and planners in the study by Hauck *et al.* (2013) identified ecosystem services maps as being useful as communication tools, for comparing different policy options, and for enhancing public acceptance by providing specific arguments for nature conservation. Albert *et al.* (2014b), who combined interviews and surveys of regional and landscape planners in Germany, reported similar findings. In discussing the importance of ecosystem services

maps as a communication tool, one interviewee said: "[…] the abstract advantages and disadvantages become graphic and consequences of planning are tangible […]."

However, decision makers are concerned that incorporating ecosystem services into spatial planning would make an already complex process even more difficult (Albert *et al.*, 2014b). Participants worried about lack of data, already cited as a problem for land-use decision making. They also highlighted the importance of recognising and quantifying uncertainty, for example, one interviewee raised concerns over: "[…] numbers that appear to be accurate but indeed are not (pseudo-accuracy)." This need for caution was also highlighted by Hauck *et al.* (2013), as one participant in that study put it: maps have "an air of authority."

Overall, more effort should be made to understand the needs of the users of ecosystem services maps and ensure that datasets are compatible with existing data and planning instruments. Ecosystem services information should be easy to use (see Box 4 for two common tools) as well as robust and transparent to stakeholders (Albert *et al.*, 2014b; Hauck *et al.*, 2013).

Box 4
Ecosystem services mapping tools

InVest, the Integrated Tool to Value Ecosystem Services and their trade-offs, is an open access tool based on land cover maps for use with GIS (Geographic Information System) software. It follows a tiered approach from simple proxy-based mapping to incorporating more complex process-based models. This tool can be used to map ecosystem services and trends over time as well as economic values of ecosystem services (Kareiva *et al.*, 2011). Other tools take a variety of different approaches, for example, SolVES, the Social Values for Ecosystem Services tool, maps perceived social, rather than economic, values for ecosystems. Such tools are costly and difficult to develop but are designed to be user-friendly for non-specialists with outputs that are understandable to all stakeholders.

2.4 Conclusions

Mapping ecosystem services is a vital part of implementing the framework; however, it is also an extremely challenging task. The natural capacity of ecosystems to provide ecosystem services, the drivers and pressures impairing that capacity, and the demand for ecosystem services all vary spatially and over time. This means there are many variables with different degrees of interdependence: a situation which calls for detailed and critical analysis.

As a result, challenges remain, in particular the need to quantify uncertainty in ecosystem services maps, a concern raised by both the academic community and policymakers (Albert *et al.*, 2014b; Schulp *et al.*, 2014). Advances in this area and others above will help to provide a powerful tool: maps which make the substantial benefits provided by the natural world explicit, allowing decision makers and the public alike to assess the true cost and benefits of different courses of action.

3. Valuation of ecosystem services

3.1 Introduction

Most people would agree that nature is 'valuable'; however, perhaps somewhat paradoxically, the idea that we place a 'value' on the natural world is a very controversial one (Gómez-Baggethun & Ruiz-Perez, 2011; Schröter *et al.*, 2014). In large part, this is because valuation is often taken to mean economic valuation. Particularly within discussion about ecosystem services, the terms valuation, economic valuation and monetary valuation are often used interchangeably (Gómez-Baggethun *et al.*, 2014; Jax *et al.*, 2013).

However, 'value' can encompass much more than economic value. It can extend to ecological, inherent, bequest, aesthetic, spiritual, health values and others (Gómez-Baggethun *et al.*, 2014). Dendoncker *et al.* (2013) state that *"valuation refers to the understanding of the worth or importance of something and may be defined as 'the act of assessing, appraising or measuring value, as value attribution, or as framing valuation (how and what to value, who values)'"*. In essence, valuation should include multiple value systems. While recognising the importance of working towards integrating diverse value systems, in this chapter we focus on the economic valuation as the most commonly used method, and then explore the emerging possibilities of a more integrated valuation.

> *'Value' can encompass much more than economic value. It can extend to ecological, inherent, bequest, aesthetic, spiritual, health values and others.'*

Opponents to the concept of economic valuation argue that it can lead to 'commodification' of nature, which opens previously non-marketed areas to market trade. This can have damaging effects, some scholars say, alienating people from nature and transforming public property and services into commodities that can be accessed only by those with purchasing power (Gómez-Baggethun & Ruiz-Perez, 2011; Robertson, 2012).

Others provide counter-arguments to these concerns, asserting that economic valuation could prove to be an important tool in protecting ecosystem services for the benefit of society (Schröter *et al.*, 2014; Atkinson, Bateman & Mourato, 2012). A key principle here is that, in the absence of economic valuation, implicitly economic-based business and political decision making will assign ecosystems a default value of zero. Against this view, appeals for conservation based on inherent and less tangible values can appear as a constraint on legitimate socio-economic progress, but this fundamentally ignores the pivotal role of ecosystems in underpinning human health and wider facets of 'quality of life'.

To reverse this misconception of nature as a constraint, rather than a fundamental resource, economic valuation could be a useful instrument in communicating the case for ecosystem service protection and accounting for 'market failures'. Many ecosystem services, especially regulating services, fall into the category of 'public goods'. These are

defined as 'non-excludable' and 'non rival', meaning that individuals cannot be effectively excluded from use, and that use by one individual does not reduce availability to others (TEEB, 2010). For example, if a forest reduces air pollution, this service cannot be parcelled up and sold to those who choose to invest in it. This results in 'market failures' meaning that landowners receive no financial rewards for providing these benefits to society and therefore have no economic justification for investing in them (Schägner *et al.*, 2013; TEEB, 2010, see Box 5).

Furthermore, the assumption that regulatory services are 'non-rival' ignores natural limits. For example, one person can 'use up' the waste assimilative capacities of air, reducing these capacities for use by others, and representing a further market failure. In addition, some economic valuation models, for example those developed by the UK National Ecosystem Assessment (2011), overlook valuation of supporting services in order to avoid the error of 'double counting', when an ecosystem service is included more than once. There is therefore an implicit assumption that their values are reflected in the contribution they make to traded services; the reality is that trading a subset of services while overlooking those supporting them lies at the heart of many sustainability problems.

Box 5
Market failures and externalities

Market failures occur because ecosystem services such as air pollution reduction or water purification are not traded and are therefore 'external' to the market. As these 'positive' externalities are not accounted for by the market, the benefits are not passed on to, for example, the owner of a forest that provides air purification for the local area. Similarly, negative externalities arise when the true cost of polluting activities is not borne by polluter. While in the past pumping polluted effluent into a river might have been thought of as a free and simple method of disposing of waste, in fact it invokes costs which are not accounted for by the markets. These can be made explicit by economic valuation (Atkinson, Bateman & Mourato, 2012; Schröter *et al.*, 2014).

As a result of the failure of the markets to account for the value of many supporting and regulating services, they have historically been neglected in decision making and thus consistently degraded, leading to progressive declines in overall system integrity, functioning and resilience (Atkinson, Bateman & Mourato, 2012). Economic valuation could help to address this problem by communicating the value of ecosystem services in comparison to man-made services, as well as making evident the true costs of activities which degrade them (TEEB, 2010).

A further benefit of economic valuation is that it can provide a single common unit which can be used to condense a complex system and to compare the impacts of alternative policy measures, a fact that could be

of great use to decision makers (Schröter *et al.*, 2014; Schägner *et al.*, 2013; Foody, 2015; Bateman *et al.*, 2014).

Ultimately, economic valuation should not be used to set a price at which to trade nature, but rather as an indication of the substantial benefits that ecosystems provide to humans, which should be considered in economic, political and ecological discourse (Foody, 2015; TEEB, 2010).

3.2 Methods of economic valuation

There is no single economic valuation technique that can be applied to all ecosystem services, as methods vary depending on the characteristics of the ecosystem services, as well as the data available (Department for Environment Food and Rural Affairs (DEFRA), 2007; TEEB, 2010). Here, we outline the methods commonly used under a Total Economic Valuation framework.

Initially, values assigned to ecosystem services can be divided into use and non-use values. Use values include direct uses such as food production, fishing or recreation. However, they also include indirect use and so-called option values. Ecosystem services which have indirect use value are often regulating services – water purification or pollination, for example – that are not directly used by humans but ultimately provide important benefits. Services providing indirect use values are often described as 'intermediate services'. Option values are linked to the potential future uses of ecosystems and biodiversity. Increased biodiversity may provide resilience in the face of global change, for instance, and preserving it now gives the option of benefitting from such a service in future. Conversely, non-use values relate to those associated with, for instance, the enjoyment provided by knowing of the existence of biodiversity, or the importance of maintaining ecosystem services for future generations (Gómez-Baggethun *et al.*, 2014).

Market prices can often be applied directly to use-value ecosystem services, such as food or timber production. This technique may need to be corrected for distortions of market value such as taxes or subsidies, but ultimately is the simplest form of economic valuation (Atkinson, Bateman & Mourato, 2012).

Some ecosystem services, often indirect-use, regulating services, are not associated with a market value directly but underpin important factors which can be valued in this way. Examples include flood protection and maintenance of agricultural productivity. These valuation techniques value ecosystem services as productive inputs and are known as 'production function' methods (Barbier, 2007).

Many ecosystem services, however, have no market prices: examples include the aesthetic beauty of a landscape or cultural significance. In this case, economists must turn to non-market valuation. The two main methods are: 'revealed preference' and 'stated preference'.

'Many ecosystem services, however, have no market prices: examples include the aesthetic beauty of a landscape or cultural significance. In this case, economists must turn to non-market valuation.'

Revealed preference methods examine the amount spent on goods related to ecosystem services which can 'reveal' how people value them. For example, recreational services might be valued based on the amount people pay to travel to an area; a park in a city might be valued based on how it increases house prices, or bird watching may be valued as the amount of bird seed bought (Clucas, Rabotyagov & Marzluff, 2014). Derived surrogate market values such as these should not be confused with the value of the thing itself, they merely reflect one facet of market value, indicating roughly how big the value is and whether it is positive or negative.

Stated preference methods often involve surveys in which people are asked how much they would be willing to pay for a service or how much they would need to be compensated for the loss of a service (Bateman *et al.*, 2010; Clucas, Rabotyagov & Marzluff, 2014). Choice modelling can also be used, where individuals are asked to choose between a range of environmental goods at different prices and their willingness to pay is calculated in this way (Alcon *et al.*, 2014).

In recent years problems with such surveys have been highlighted (Parks & Gowdy, 2013). These methods rely on the individual as a rational being, despite numerous psychological experiments showing that this is not the case. One pertinent example here is that people tend to feel that a loss matters more than a gain of the same magnitude. As a result, willingness to pay for a gain is often lower than willingness to accept a loss. The current destruction of the natural world suggests that willingness to accept a loss is more relevant now, however most of these kinds of valuation exercises rely on willingness to pay, with the possible side effect that such losses are valued lower than they should be (Parks & Gowdy, 2013). People also often exhibit different preferences depending on the context: for example, Bateman & Mawby (2003) found that willingness to pay for environmental benefits was higher if the interviewer wore more formal clothing.

Stated preference methods are often the only option for the ecosystem services that are the most difficult to value, including many non-use services such as cultural identity or heritage values (Chan *et al.*, 2012). Unfortunately, research shows that these methods are likely to work best when individuals have a good knowledge of, and clear preferences for the ecosystem services being valued, which is often not the case for cultural services such as existence values (Atkinson, Bateman & Mourato, 2012; Bateman *et al.*, 2010).

A further difficulty arises from survey methods that implicitly assume that an aggregate of individual valuation is representative of society-wide value. Often, people develop different values when deliberating with others, revealing shared values reflective of what an ecosystem means to a community as a whole (Fish *et al.*, 2011).

Together, these methods demonstrate the intricacies of economic valuations of different types of ecosystem services. However, to be of real use to policymakers, valuations need to be mapped. This presents further challenges, as both demand and supply of ecosystem services may vary spatially, leading to significant variations in economic value (Schägner *et al.*, 2013). Furthermore, in order to evaluate the success of policy measures, such maps must also have a temporal dimension, so that changes over time can be assessed (Atkinson, Bateman & Mourato, 2012).

The most common method of creating maps of economic value, especially over large scales, is the value transfer method (Brouwer, 2000). This involves using data from a site or sites where the economic value of ecosystem services has been assessed and transferring this to larger scales (Troy & Wilson, 2006; Brouwer, 2000). This method is often used because it is too costly and difficult to collect first-hand data across very large areas.

Value transfer analysis can be based on one of several methods:

i. *Unit values.*
ii. *Adjusted unit values.*
iii. *Value functions.*
iv. *Meta-analytic value functions.*

Unit values are the simplest of the four approaches and are based on a value per unit of ecosystem services; the spatial variation is therefore mapped only as variation in supply in ecosystem services, with the value of a single unit of ecosystem services assumed to be constant across space.

In *adjusted unit value* analysis other variables, such as population density or income levels, are used to spatially adjust the values. The *value function* method is yet more complex, including multiple spatial variables and accounting for how these influence value. The 'function' – or mathematical summary of relationships – is drawn up by an intensive study of a site and is then applied to the rest of the mapped area. Similarly, for *meta-analytic value function*, a value function is created, however, in this case it is based on a meta-analysis, using results from a range of previous studies to define the relationships within the function (Johnston & Rosenberger, 2010). Brander *et al.*(2012), for example, quantified the economic impacts of climate change on European wetlands using a value function based on a meta-analysis of 222 studies.

In their review Schägner *et al.*(2013) found that unit values were by far the most common method, used by 78% of the 79 studies they examined. Value functions were used by 20%, adjusted values by 5% and just 4% used meta-analytic value functions.

However, the frequency of use of these methods does not tell us which is the best, or the most reliable. Errors can arise in valuation maps for several reasons. Firstly, there may be inaccuracies in estimates of ecosystem services supply (discussed in Chapter 2) or ecosystem services value (discussed above in Section 3.2). Secondly, all value transfer methods are prone to generalisation errors. These arise as a result of using data from a single site or sample of sites to generalise about value on a larger scale.

For example, Boyles *et al.* (2011) estimated the value of agricultural pest control by bats in the US as US$ 22.9 billion per year. Fisher & Naidoo (2011), however, expressed deep concerns about the analysis, pointing out that it was extrapolated from specific sites in Texas but that crop mixtures, market prices, pests and bat feeding ecology all vary substantially across the US, with substantial impacts on economic value.

It is for this reason that meta-analytic value functions, despite being rarely used, may be the most accurate (Johnston & Rosenberger, 2010; Schägner *et al.*, 2013). Like value functions, meta-analytic value functions can take into account multiple variables but the relationships between these are based on more extensive, primary research. The added accuracy of this is, of course, dependent on the quality of the data that the meta-analytic value function is drawn from (Schägner *et al.*, 2013).

A further cause for concern is that the simpler methods, such as unit values, tend to be used when mapping multiple ecosystem services, with the more accurate meta-analytic value functions mainly used to map a single service (Schägner *et al.*, 2013). This could be problematic, because, as discussed throughout this report, it is vital that trade-offs and synergies among ecosystem services – and hence economic values – are apparent to policymakers.

Indeed, a key gap in the research is the lack of information regarding the uncertainty surrounding economic estimates (Schägner *et al.*, 2013). As discussed in Chapter 2, such information is of paramount importance to policymakers, it can tell them whether the map gives a rough indication or provides an accurate reflection. This can make a vital difference to how such maps are used in decision making. Despite recognition of this, few studies attempt to quantitatively assess error (Schägner *et al.*, 2013).

> *'A key gap in the research is the lack of information regarding the uncertainty surrounding economic estimates.'*

Overall, there are several issues with current techniques of economic valuation of ecosystem services that require further research and discussion. As well as the lack of recognition and quantification of uncertainty in valuation, techniques for assessing and monetising non-use values have been widely criticised (Parks & Gowdy, 2013; Bateman & Mawby, 2004; Gómez-Baggethun *et al.*, 2014; Chan *et al.*, 2012). Cultural ecosystem services in particular have been neglected by economic valuation studies (Pröpper & Haupts, 2014; Chan *et al.*, 2012). Similarly, option values, such as protection of biodiversity to provide resilience in the face of global change, should be included in any valuation of our natural capital but have as yet been neglected. Failure to account for these factors because of the difficulty associated with their economic valuation may exclude these important issues from the decision-making process (Chan *et al.*, 2012).

Discount rates

An important aspect of valuation, especially when considering protecting natural capital for future generations, is discount rate: the amount that a benefit declines in value each year into the future it extends. Discount rates are made up of two components: time preference and opportunity costs. Time preference refers to how much we prefer to reap benefits now, compared to in the future. For example, faced with the choice of €70 now or €100 in 10 years, many will choose the instant returns, even though they are lesser. Opportunity costs refer to the opportunities missed by locking away that investment for 10 years. For example an individual may invest €100 in their car now, to prevent €100 worth of damage in a year. However, if they had invested that money elsewhere – in a savings account with an interest rate of 3%, for example – they would have €103 after a year. The opportunity costs are therefore €3 and it is only worth €97 to protect the car against €100 worth of damage in a year (Roberts, 2012). If we set the discount rate high we are indicating that the future value of ecosystems declines quickly. The higher we set the discount rate the less it is worth to us now to invest in protecting ecosystems for the future (Dasgupta, 2008).

How to set a discount rate is highly contentious (Carpenter *et al.*, 2009). Some argue that discount rates should be calculated from time preferences set according to actual investment behaviour, and opportunity costs based on interest rate. In essence, this implies that we should only invest in natural capital if it offers better returns that putting the money in a bank. However, this assumes that we can replace an ecosystem and the benefits it provides by using money, and this is, of course, very far from the truth. In addition, researchers have pointed out that how people choose to invest their money individually may not be the same as what they feel we owe to future generations (Kelleher, 2012). Setting a discount rate is a moral judgement that cannot be the province of policy-makers and economists alone, but is an issue for the whole of society.

3.3 Use of economic valuation

A central question for those involved in research into the best methods of economic valuation is: how will it be used by policymakers? Although there is much analysis and discussion in the academic literature regarding methodology, research into how such valuations are actually used in decision making is scarce, and this can hamper attempts to improve both techniques and outcomes (Laurans *et al.*, 2013). Laurans & Mermet (2014) suggest that the importance of economic valuations lies in three key points:

i. Helping to rationalise the decision making process.

ii. Offering a method of comparing and optimising policy outcomes.

iii. Helping to frame the process, by providing a way of organising disparate types of information.

Here, we explore two key areas where economic valuations of ecosystem services could provide important support to policy makers.

3.3.1 National Accounts

Gross domestic product (GDP) provides only a limited representation of the wealth of a country. Rather than provide a balance sheet, as is standard for any business, it is based purely on income, and does not show the assets a country already has, including its 'natural capital', which may provide significant wealth (see Figure 8 on page 22). Furthermore, not only does GDP omit natural capital, it neglects to account for the fact that some income-generating activities may degrade natural capital. Therefore, as GDP grows, natural resources may be in decline as a direct result, ultimately reducing the true wealth of a country. Thus, GDP as it is currently measured provides not only a limited representation of the wealth of a country but a distorted one (Patil, 2012; Costanza & Talberth, 2009).

In recognition of the importance of this issue, the UN Statistical Commission adopted the System for Environmental and Economic Accounts in 2012, which provides a method to account for material natural resources like minerals, timber and fisheries. However, it does not include other ecosystem services that do not have a market price. As discussed above, accounting for such ecosystem services is crucial because they include regulating services which may underpin many others (Everard, 2014a).

The UK National Ecosystem Assessment Follow-on programme presents a case study of economic valuation at the national scale, highlighting the importance of including non-market values. The study assessed the economic costs and benefits of planting 15 000 hectares of new woodland per year from 2014 to 2063 in Great Britain. Using models which considered only market values, the results showed that the costs, at £79 (€106) million, outweighed the benefits of £65 (€88) million. In contrast, when the models were designed to include non-marketed goods, such as carbon storage and recreation, although the costs were higher, at £231 (€311) million, they were outweighed by benefits of £546 (€736) million. The higher costs are the result of displacing profitable agricultural land with woodland, but this was necessary to realise the social benefits because it redistributed woodlands closer to urban centres, where more people could benefit from them (Bateman *et al.*, 2014).

3.3.2 Policy instruments

One of the main aims of carrying out economic valuations of ecosystem services is to ensure that externalities in the form of costs of negative environmental impacts and benefits of healthy ecosystems are accounted for. Ensuring that environmental externalities are not neglected is also the goal of many policy instruments used to drive sustainability, such as taxes or subsidies, and ecosystem services valuations could be useful for their effective design. Such instruments can take the form of 'polluter pays' whereby the costs of negative environmental impacts are charged to the perpetrator. Schemes based on this concept include Europe's central climate policy instrument, the
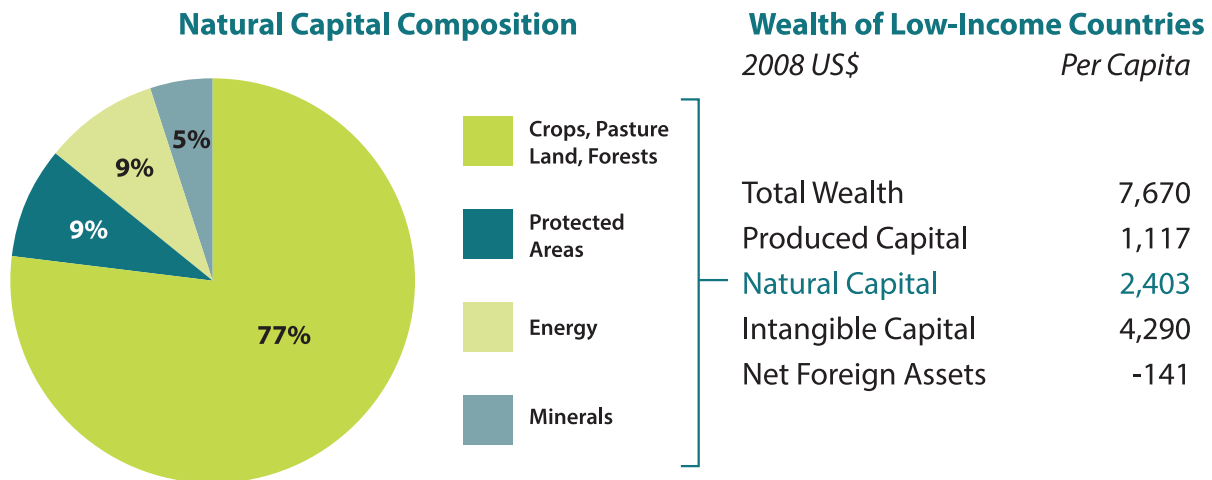
Natural Capital Composition

Wealth of Low-Income Countries
2008 US$ *Per Capita*

Crops, Pasture Land, Forests	
Protected Areas	
Energy	
Minerals	

Total Wealth	7,670
Produced Capital	1,117
Natural Capital	2,403
Intangible Capital	4,290
Net Foreign Assets	-141

Figure 8. Reprinted from World Bank Environment Brief on Natural Capital Accounting. The composition of natural capital of Low-Income Countries[3]. Retrieved from http://www.worldbank.org/en/topic/environment/brief/environmental-economics-natural-capital-accounting

EU Emissions Trading Scheme. This scheme sets a cap on the amount of CO_2 that can be released overall but within this allows polluters to trade carbon allowances[4].

Where 'polluter pays' schemes recognise negative environmental impacts that are not accounted for by the markets, instruments based on the 'steward earns' principle provide rewards for protection or restoration of ecosystems and their services (Gómez-Baggethun & Ruiz-Perez, 2011; Vatn, 2010). These schemes are generally known as 'payments for ecosystem services' (PES).

Costa Rica was an early pioneer of PES, implementing nationwide schemes since 1997 to conserve forest by payments for protection, reforestation, sustainable management and regeneration (Porras *et al.*, 2013). A high profile, international example of PES is the 'reducing emissions from deforestation in developing countries' or REDD[5] scheme which has been under negotiation by parties to the United Nations Framework Convention on Climate Change since 2005. Under the REDD scheme, richer nations pay into a fund that is then used to reward developing countries for efforts to reduce emissions and enhance removals of greenhouse gases through various forest management options. In Europe, agri-environment schemes provide a long running example of PES. First implemented in the 1980s, under these schemes farmers are paid to carry out various activities which enhance and protect ecosystem services and biodiversity, such as preserving hedgerows or leaving unmown strips of habitat for wildlife[6].

The name 'payments for ecosystem services' would suggest that economic valuation of ecosystem services would be essential to the design of such schemes. However, the amounts paid out are generally related, not to the value of the ecosystem services provided, but to the cost of actions carried out by 'stewards', such as farmers in the case of agri-environment schemes (Reed *et al.*, 2013; Wunder, 2013). In practice, most PES schemes include an element of faith that actions

will be likely to produce results which may accrue only after a long time and at an uncertain level (Smith *et al.*, 2013).

Reed *et al.* (2013) highlight the fact that economic valuations could be of real use to improve the efficiency of such schemes. Using PES for peatland ecosystem services as a case study, they suggest a shift from 'payments for actions' to 'payments for results', i.e. payments for measurable improvements in ecosystem services. Although payments would need to account for both costs to stewards and benefits to society, incorporating uncertainties and long-term outcomes, spatially explicit maps of the economic value of ecosystem services and of costs of restoration could provide essential information on where ecosystem services can be most efficiently provided (Reed *et al.*, 2013).

3.4 Beyond economic valuation

At the beginning of this chapter we referred to the argument that one of the main benefits of monetary valuation is that it can provide a single, common unit which can be used to condense a complex system (Bateman *et al.*, 2014; Schägner *et al.*, 2013). However, some scholars have argued that it is not possible to reduce nature's 'value' into a single, fundamental unit (De Groot, Wilson & Boumans, 2002; Norton & Noonan, 2007; Parks & Gowdy, 2013). An alternative approach has been proposed: 'value pluralism' which suggests that there are several distinct values, which may be in conflict with each other but which are all of equal importance and cannot be reduced to a unique value (Gómez-Baggethun & Ruiz-Perez, 2011; Dendoncker *et al.*, 2013; Jax *et al.*, 2013; Gómez-Baggethun *et al.*, 2014). This approach may be more difficult to conceptualise and develop but it reflects reality, its proponents argue: people do hold a diverse range of values (Norton & Noonan, 2007).

Although valuation techniques are still under discussion in the EU, the conceptual framework for EU-wide ecosystem assessments makes the importance of both monetary and non-monetary values explicit (Maes

3. http://www.worldbank.org/en/topic/environment/brief/environmental-economics-natural-capital-accounting
4. http://ec.europa.eu/clima/policies/ets/index_en.htm
5. http://www.un-redd.org/
6. http://ec.europa.eu/agriculture/envir/measures/index_en.htm

et al., 2013a, see Figure 1 on page 4). If ecosystem services valuation is to take this plural road, however, the challenge of creating an integrated valuation system will be need to be faced. Work on this has already begun: the Ecosystem Services Partnership has set up a thematic working group on value integration and the EU-funded OpenNESS[7] project has published a report on developing an operational, integrated valuation framework (Gómez-Baggethun *et al.*, 2014).

> *'[Integrated valuation] includes two sets of values, cultural and ecological, which run alongside monetary values.'*

The OpenNESS report defines integrated ecosystem services valuation as: *"the process of synthesizing relevant sources of knowledge and information to elicit the various ways in which people conceptualize and appraise ecosystems services values, resulting in different valuation frames that are the basis for informed deliberation, agreement and decision"* and includes two sets of values, cultural and ecological, which run alongside monetary values (see Figure 9). The report stresses that the idea of *integration* of values is key; rather than a series of different types of ecosystem service valuations, integrated valuation must also show how these values relate to each other (Gómez-Baggethun *et al.*, 2014). For instance, which values are in conflict and which serve to reinforce each other?

In working towards an operational framework for integrated ecosystem services valuation they recommend four basic steps:

i.   The purpose of valuation should be defined as soon as possible in the process.

ii. A scoping study should follow to identify different values, knowledge systems and information sources. This may include a deliberative process to address the breadth of societal values, including those that may be more deeply held, shared across communities and less immediately evident.

iii. The relevant valuation techniques should be chosen based on the purpose of valuation and the scoping studies.

iv. How the values relate to each other, including synergies and conflicts should be examined.

3.5 Conclusions

Economic valuation of ecosystem services can provide a useful tool to policymakers. Firstly, in raising awareness regarding the substantial benefits that ecosystems provide. *Nature is valuable* may be a statement that many people agree with in a vague, general sense, but the statement *Planting 15 000 hectares of woodland boosts Britain's economy by £315 million* (Bateman *et al.*, 2014) quantifies this value and makes it explicit in terms that all stakeholders will be familiar with. Secondly, it can help to target resources, to provide the most efficient protection of ecosystems and their services with the limited funds available (Glenk *et al.*, 2013; Bateman *et al.*, 2014). Thirdly, it can help to rationalise and frame the decision-making process, providing points for further discussion and deliberation (Laurans & Mermet, 2014).
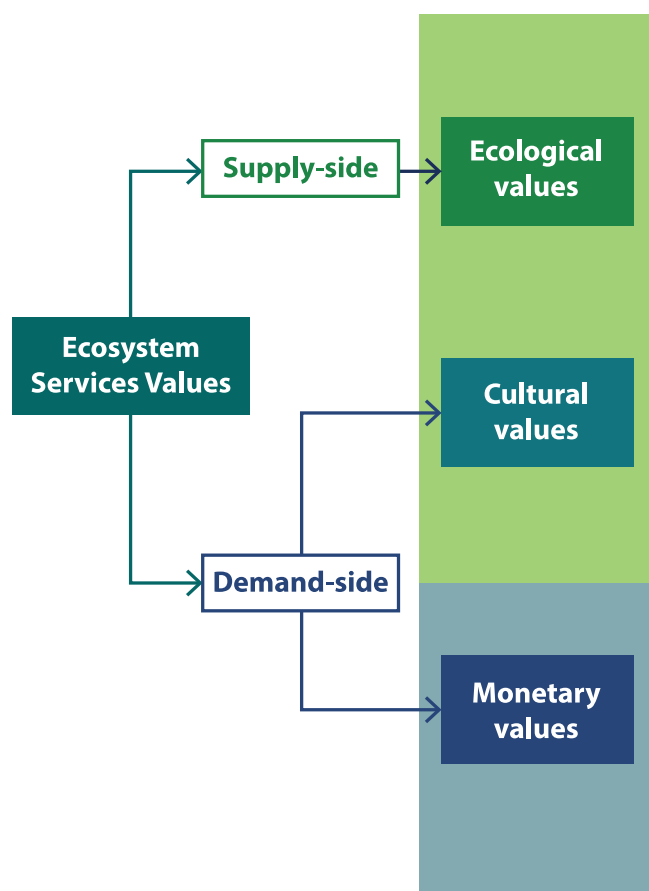

Figure 9. Adapted from (Gómez-Baggethun *et al.*, 2014). Integrated valuation of ecosystem services considers non-monetary and monetary valuation methods and value pluralism.

Economic valuation also has significant drawbacks. Firstly, quantification of uncertainties surrounding valuations and value mapping is often neglected, yet this is key information for any policy decision (Schägner *et al.*, 2013). Secondly, economic valuation techniques to account for non-use values are weak (Parks & Gowdy, 2013; Chan *et al.*, 2012). This is worrying as it could mean that these important aspects are excluded from decision making. Thirdly, reliance on economic valuation assumes that all values can be condensed down to a fundamental, monetary value. However, it is now increasingly recognised, in both academic research and policy literature, that there are multiple values, all of equal importance (Norton & Noonan, 2007; Gómez-Baggethun *et al.*, 2014; De Groot, Wilson & Boumans, 2002; Parks & Gowdy, 2013).

Economic valuation has a role to play, but it must be considered alongside other types of value. Through the use of integrated valuation, the benefits of economic valuation – awareness raising, resource targeting and process framing – can all be enhanced. Conversely, the problems, such as failure to adequately account for non-use values, can be dealt with. The challenge now lies in producing an integrated valuation framework that can be readily used by policymakers, but we are already making progress towards this.

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7. OpenNESS (Operationalisation of natural capital and ecosystem services) was funded by the European Commission under the 7th framework. See: http://www.openness-project.eu/

4.   The importance of systems thinking

4.1 Introduction

A theme running through this report has been the importance of considering ecosystem services as part of a wider system. We have discussed the need to account for multiple services, supply and demand, short and long-term supply, the links between biodiversity and ecosystem services and resilience (Everard & McInnes, 2013; Raffaelli & White, 2013; Schröter *et al.*, 2014; Bennett, Peterson & Gordon, 2009).

'Systems thinking' is an acknowledgement that *"everything affects everything else in the natural world"* (Raffaelli & White, 2013). Considering any part in isolation, and failing to account for its impacts on the rest of the system, can therefore have unforeseen and possibly damaging consequences (Maestre Andrés *et al.*, 2012). An overwhelming focus on maximising provisioning services, for example, is thought to be the largest driver of biodiversity loss over the last 50 years, with the result that a wide range of non-marketed ecosystem services, including genetic resources and pollination, have been significantly reduced across the globe (MA, 2005).

> *'An overwhelming focus on maximising provisioning services is thought to be the largest driver of biodiversity loss over the last 50 years.'*

When examining how different elements of the system are interconnected, widely used terms within ecosystem services research include: 'trade-offs', 'synergies' and 'ecosystem service bundles'. Here, we explore exactly what these mean in the context of ecosystem services and their importance to practical decision making.

4.2 Trade-offs, synergies and ecosystem services bundles

Trade-offs are commonly defined as an increase in one ecosystem service resulting in a reduction in another (Rodríguez *et al.*, 2006). For example, felling a forest to grow corn maximises food provision but reduces carbon storage, storm buffering, air quality and flood regulation. One service is therefore 'traded off' against others. Spatial trade-offs occur when maximising one ecosystem service reduces another in a different location. An example of this is the hypoxic or 'dead' zone in the Gulf of Mexico that has been so heavily polluted by fertiliser run-off it can no longer support marine life. In this spatial trade-off, maximising provision of crops has been traded-off against fishing catch, recreation, system resilience, etc. in another location (Rodríguez *et al.*, 2006).

Trade-offs can also occur over time. Maximising an ecosystem service in the short term may reduce its supply in the long term (Mouchet *et al.*, 2014). Again, examples of this include some intensive agricultural practices, which may maximise crop production in the short term but

have negative effects on soil structure and fertility, genetic resources and soil erosion, causing yields to decline in the long term.

Trade-offs may also refer to the beneficiaries of ecosystem services. For example, a trade-off can occur between the users of an ecosystem service, when 'winners' reap the benefits of an ecosystem service but 'losers' bear the costs (Mouchet *et al.*, 2014; Howe *et al.*, 2014). A river, for instance, could be diverted to provide water for irrigation, benefitting farmers from one region but impacting the livelihoods of fishers downstream.

Finally, one important aspect of all trade-offs is whether they are reversible; some ecosystem services will be able to return to their original levels after the negative impacts of a trade-off, but this is not always the case (Mouchet *et al.*, 2014). For example, when fisheries are harvested at or below Maximum Sustainable Yield (MSY), they retain a capacity for regeneration and for full recovery of fish populations when exploitation ceases or reduces, whereas fishing above MSY is likely to result in degradation of fish stocks and eventually to the irreversible demise of the fishery (Appleby *et al.*, 2013).

Synergies occur where increases in one service are coupled with increases in another. An obvious example of a synergy occurs between the regulating service pollination and the provisioning service crop production. In fact, 75% of the world's major crops are dependent on, or benefit from pollination (Carvalheiro *et al.*, 2012). There is also a synergy between soil erosion control and crop production. Erosion can result in a loss of the more fertile soil, reducing yields. Good erosion control can therefore mean better supply of crops (Bennett, Peterson & Gordon, 2009). If soil erosion prevention measures involve planting or protecting vegetation along river banks this can also boost water purification, creating a further synergy (Gundersen *et al.*, 2010).

Recent research by Jopke *et al.* (2015) developed an easily understandable statistic, R, for ranking ecosystem services according to their synergistic value. The higher the R of an ecosystem service, the more synergies and fewer trade-offs it has with others. Using data on 10 ecosystem services across the EU they found that crop capacity had the lowest R (as it was associated with the most trade-offs relative to synergies) and carbon storage the highest, as it had the most synergies in comparison to trade-offs.

Finally, ecosystem service bundle is a term used for sets of ecosystem services that repeatedly occur together across space or time (Raudsepp-Hearne, Peterson & Bennett, 2010). Queiroz *et al.* (2015)[8] identified five ecosystem services bundles in a region of Sweden. For example, one bundle was made up of the regulating services water quality, and nitrogen and phosphorus retention; the provisioning service of timber; and the cultural service of moose hunting. These were all spatially clustered in the landscape, in areas dominated by forest but with scattered towns. Identification of ecosystem services bundles may be useful to help decision makers account for 'multi-functionality' of landscapes and improve management (Raudsepp-Hearne, Peterson & Bennett, 2010; Kareiva *et al.*, 2007).

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8. *This work was conducted as part of the Program on Ecosystem Change and Society (PECS). This is a 10-year project funded by the International Council for Science and UNESCO which aims to aims to integrate research on the stewardship of social–ecological systems.*
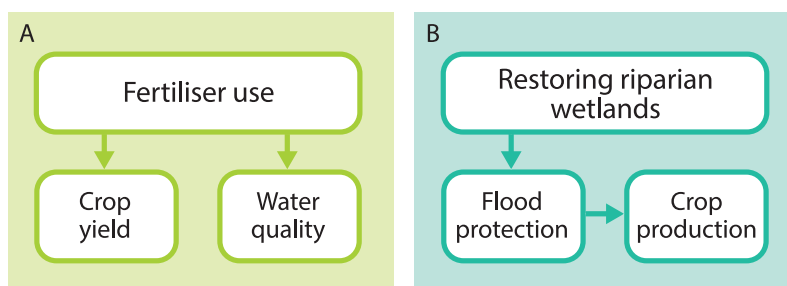
Figure 10. Adapted from Bennett, Peterson & Gordon (2009). In panel A the ecosystem services of crop yield and water quality have a shared driver, fertiliser use, but no direct impact on each other. In panel B the driver, wetland restoration, affects only one ecosystem service, flood protection, but this can directly affect another ecosystem service, crop production.

4.3 What drives ecosystem services associations?

Understanding what causes the observed trade-offs and synergies between ecosystem services is vital for good decision making and management. Such associations can be the result of two factors:

i. A shared driver affecting multiple ecosystem services.

ii. Interactions between the ecosystem services themselves (Bennett, Peterson & Gordon, 2009).

Fertiliser use, for example, is a driver that positively affects crop production but negatively effects the provision of good quality water (panel A of Figure 10). This can create a trade-off between the two services, although there is no direct interaction. Direct interactions between ecosystem services can be both positive and negative. Flood protection, for instance, can prevent farmland from becoming waterlogged and therefore have a positive effect on crop production (panel B of Figure 10). Carbon sequestration can have a negative effect on the quantity of water provided, as greater carbon sequestration requires more plants, taking up more water.

Research into the associations between ecosystem services and the identification of ecosystem service bundles is now increasing (Howe et al., 2014; Queiroz et al., 2015; Briner et al., 2013; Lauf, Haase & Kleinschmit, 2014). Howe et al. (2014) reviewed studies from 2000–2013 on this topic, with a total sample of 231 trade-offs and synergies. They found that trade-offs were much more common, being reported three times as often as synergies.

The researchers also explored the conditions that generally lead to a trade-off. Firstly, the factor with the greatest influence was whether the beneficiaries of the ecosystem services held a private or public interest (i.e. whether or not the ecosystem services involved are 'public goods'). A trade-off occurred in 81% of cases involving stakeholders with a private interest. Secondly, trade-offs overwhelmingly involved provisioning services; only 5% of trade-offs did not include a provisioning service. While the authors concede that only a few studies did not include provisioning services (11) they note that seven of these (64%) resulted in a synergy. This tendency is perhaps unsurprising, given the evidence pointing towards the damaging effects of maximising provisioning services without consideration of other services (MA, 2005).

The authors also found that, in terms of ecosystem services beneficiaries, trade-off 'winners' were significantly more likely to hold a private interest and use provisioning services. In 69% of trade-offs, winners held a private interest, used provisioning services and acted at local rather than regional or global scales. In the same cases, over 90% of losers held a public interest and used regulating, cultural or supporting services (Howe et al., 2014).

4.4 Lessons for management and policy making

Any change in land use or management will influence the overall properties and functioning of the system and can therefore alter supply, not only of a single ecosystem service, but of an entire suite of services (de Groot et al., 2010). Understanding trade-offs and synergies allows policymakers to reduce the damaging effects of focusing on a few services at the expense of others (Rodríguez et al., 2006).

Everard & McInnes (2013) draw a distinction between management and decision making that 'optimises' multiple services, providing greatest net value and resilience to society and the still-prevalent tendency to 'maximise' one or a few services, often resulting in net societal detriment.

A real-world example of optimising versus maximising is given by Queiroz et al. (2015) who found a general absence of the expected trade-off between agricultural provisioning services and regulating services in the Stockholm region of Sweden. The study did show, however, that the hotspots of agricultural production were coldspots for regulating services. This demonstrates that the low-intensity farming practices generally found in the region are able to optimise several different ecosystem services, but in the small areas where agricultural production is maximised other services suffer as a result.

Creating synergies is often thought to be the ultimate goal for managers and policymakers, however, while certainly desirable, research shows that policymakers must not lose sight of the importance of dealing with trade-offs (Goldstein et al., 2012; Howe et al., 2014). To begin with, although more research is needed into synergies, current evidence suggests that trade-offs are more common (Howe et al., 2014). Furthermore, in their review, Howe et al. (2014) found that in cases where synergies between ecosystem services had been successfully fostered, this was primarily the result of avoiding or overcoming the

'Understanding trade-offs and synergies allows policymakers to reduce the damaging effects of focusing on a few services at the expense of others.'

main reasons that trade-offs occur. These included:

i.   Failure to account for all benefits or stakeholders.

ii. Failed management and the damaging effects of pollution and/or habitat destruction.

iii. An assumption that provisioning services should always dominate other services.

These factors should be explicitly considered and addressed by decision makers as early as possible (Howe *et al.*, 2014).

Applying a systemic approach also requires the recognition of the complexities of the system within policy making itself. A multi-sectoral perspective is needed, shifting policy away from isolated 'silos' where, for example, a budget for improving water quality might be dealt with entirely separately from that of soil erosion control, missing opportunities to foster positive outcomes for both (Harrison *et al.*, 2010; Gale, 2000). As Everard & McInnes (2013) describe, trade-offs among ecosystem services can occur not just across space and time, but also across organisational structures.

Furthermore, participation of a wide array of stakeholders including scientists, NGOs, ecosystem services beneficiaries, planners, and policymakers may all be needed to reflect the wider system (Everard & McInnes, 2013; Martínez-Harms *et al.*, 2015). In their ecosystem services decision framework (see Figure 11), Martínez-Harms *et al.* (2015) point to the importance of the social-ecological context, which should involve stakeholder preferences as well as incorporating broader institutional, governance and legal regimes.

By its very nature, a systemic approach encompasses complexity. This may make the challenges of implementing such an approach appear overwhelming. However, a useful concept here for decision makers is that of 'anchor services' (*sensu* Everard, 2014b). The principle of an 'anchor service' is that there will always be a primary need – for example, flood management – driving investment in a scheme. The systemic approach can then inform the key questions:

i.   What other ecosystem service co-benefits can flow from this intervention?

ii. What unintended ecosystem service consequences can be avoided?

4.5 Conclusions

There has been some concern that the concept of ecosystem services, while providing an effective tool to communicate the importance of what nature does for us, is too simplistic, and will blind us to the true complexities of the system (Fu *et al.*, 2013; Norgaard, 2010). Conversely, concerns have been expressed that implementing this 'simplistic' system entails excessive complexity. These conflicting perspectives are indeed a source for concern, because, as detailed here, it is vital that those complexities are recognised and accounted for, rather than overlooked at net cost to society and the ecosystems that sustain it. However, this is not a problem with the ecosystem services concept *per se*; rather, it is a misuse of it (Schröter *et al.*, 2014).

Taking a systems approach can and should lie at the heart of the ecosystem services concept. It is this transition, from narrow framing of problems and solutions towards addressing systemic implications, which signals a significant change in the culture of decision making. Decision makers must be aware that maximising one ecosystem services in the short term could have damaging effects on other services, as well as on long-term provision and resilience. We have seen that particular care must be taken to ensure that private financial gains do not result in widespread social losses, and that all efforts should be made to optimise the entire system for the greatest benefits to society. These are issues that require greater coherence between agreements at international and national scales and, at the local scale, between the practical factors shaping the decisions of resource owners and managers (Everard *et al.*, 2014c).
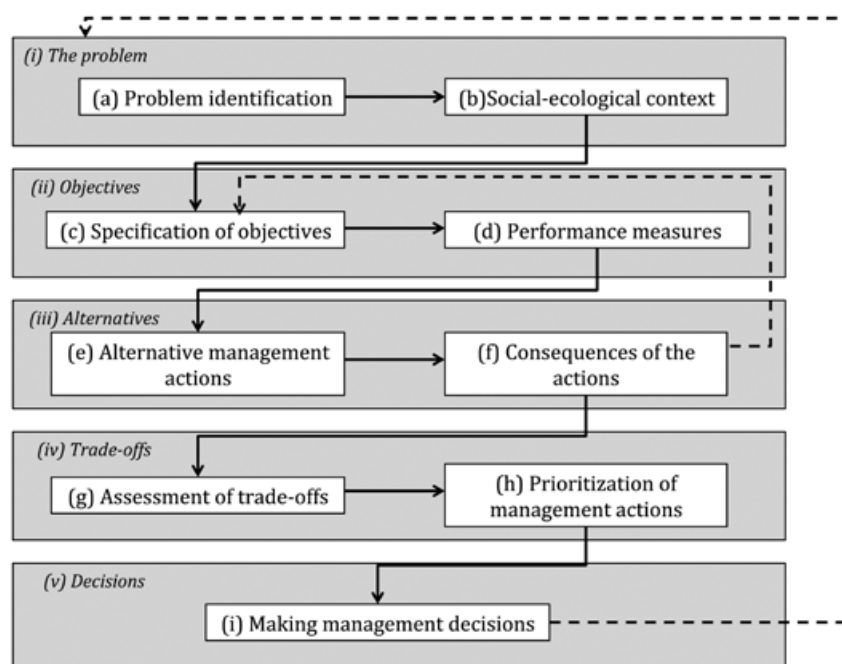

Figure 11. From Martínez-Harms *et al.* (2015), showing the core steps in the ecosystem services decision-making process. The inner dotted line shows the evaluation of the outcomes of the alternative management actions against the objectives, and indicates the possibility to update the objectives. The outer dotted line shows the process of 'adaptive management' whereby the outcomes of an action are monitored and assessed and remaining problems can then be addressed by following the steps again.

5.   Conclusions

The ecosystem services concept now lies at the heart of many policies and initiatives designed to protect biodiversity (Maes *et al.*, 2013b). Concerns have been raised, however, that a focus on protecting ecosystem services may only protect a subset of biodiversity, with the result that it continues to decline overall, irreversibly impoverishing our planet (Schröter *et al.*, 2014).

There is now a consensus within the scientific community that biodiversity has a fundamental role to play in ecosystem functioning, underpinning essential processes such as resource capture, biomass production and nutrient recycling (Cardinale *et al.*, 2012). Although research into the links between biodiversity and ecosystem services is less developed, there is mounting evidence that biodiversity is also vital for ecosystem services provision (Cardinale *et al.*, 2012). While there may be specific ecosystem services for which biodiversity is not a key component in the short term – crop production, for example – there is evidence to show that it is likely to be crucial for maintaining the stable provision of multiple ecosystem services, in the long term and under global environmental change (Gamfeldt *et al.*, 2013; Balvanera *et al.*, 2014; Cardinale *et al.*, 2012; Isbell *et al.*, 2011; Naeem, Duffy & Zavaleta, 2012).

Stable, long-term supply of ecosystem services and resilience in the face of environmental upheaval is, of course, an important part of protecting ecosystem services for future generations. We therefore neglect biodiversity protection at grave risk, even if we do not yet know of a 'purpose' for all of it. Policies to monitor and protect ecosystem services should not replace those designed to monitor and protect biodiversity. Rather, such approaches can work alongside each other with very likely synergistic outcomes (Reyers *et al.*, 2012).

The need to consider multiple ecosystem services as well as biodiversity and resilience in decision making adds complexity; however, mapping of ecosystems and their services – a key tool for policymakers – is now gathering pace (Maes *et al.*, 2012a; Malinga *et al.*, 2015; Martínez-

'We therefore neglect biodiversity protection at grave risk, even if we do not yet know of a 'purpose' for all of it.'

Harms & Balvanera, 2012). Ecosystem services maps are increasingly sophisticated, with simple land cover maps being augmented with detailed primary data as well as process-based models that can be used to predict the impact of future changes on ecosystem services supply. More mapping studies are now considering multiple ecosystem services across different scales, with important insights for policymakers (Queiroz *et al.*, 2015). The key challenges of data availability and accounting for uncertainties have been now been identified and increasing effort will doubtless be directed to addressing them in the future (Maes *et al.*, 2012a).

Alongside mapping and assessment, economic valuation is also presented as a tool to aid implementation of the ecosystem services concept. Proponents argue economic valuation can help to target resources; raise awareness of the substantial benefits provided by ecosystems; and provide the starting point for further deliberation (Glenk *et al.*, 2013; Bateman *et al.*, 2014; Laurans & Mermet, 2014). Conversely, critics say it can lead to 'commodification' – setting a price at which to trade nature – with potentially devastating environmental and social impacts (Gómez-Baggethun & Ruiz-Perez, 2011; Raffaelli & White, 2013). Many scholars also agree that uncertainties in economic valuations are not sufficiently quantified, and that techniques to account for non-use values are weak, potentially excluding a host of services, especially cultural services, from the decision-making process (Parks & Gowdy, 2013; Chan *et al.*, 2012; Schägner *et al.*, 2013).

Integrated valuation may provide a solution which retains the benefits of economic valuation while compensating for the weaknesses. This system accounts for a variety of different values under three broad categories: ecological, cultural and monetary. Integrated valuation recognises that there are multiple values, all of equal importance and that these cannot be reduced to a single fundamental unit, but must be considered simultaneously. Although this system appears challenging to incorporate into policy making, progress towards a clear operational framework for integrated ecosystem services valuation has already been made by the EU OpenNESS project (Gómez-Baggethun *et al.*, 2014).

Finally, a key thread running through this entire report is the importance of recognising that biodiversity, ecosystem services and people are all part of a single, interconnected system. Decision makers should be vigilant against short-sighted failures to account for the damaging effects of maximising one ecosystem services at the expense of others, or trading off long-term provision against short-term gain. Nor can we afford to lose sight of the importance of biodiversity, both for its inherent value and in providing resilience and stability in the supply of ecosystem services.

The need to consider the entire, interlinked socio-economic-ecological system may seem daunting, but this is where the strength of the ecosystem services concept lies. Although the challenge is substantial, the benefits are great, and work has already begun.

Old beech forest in spring, Germany. ©iStock.com/Sabine Hortebusch

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